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Eutrophication of Lake Ray Hubbard, Dallas, Texas

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Elaine Meckel, Magda Abdul-Rahman, and G. Fred Lee.

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EUTROPHICATION OF LAKE RAY HUBBARD
DALLAS, TEXAS

G. Fred Lee, Magda Abdul-Rahman, and Elaine Meckel
With the Assistance of
Mike Gary, William Cale, Walter Rast
Douglas B. Cargo and R. Anne Jones

Occasional Paper Number 15
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PREFACE

The original impetus for initiation of this project was developed by Dr. G. Fred Lee while at the University of Texas at Dallas (UTD). It was based on an observation that the numerous water bodies in north central Texas all seemed to be showing advanced stages of eutrophication. It was found by Dr. Lee that there were essentially no previous studies on the eutrophication of southern water bodies. In the past, essentially all studies devoted to understanding eutrophication of lakes and impoundments were conducted on north temperate water bodies.

Public Law 92-500, the 1972 Amendments to the Federal Water Pollution Control Act, Section 314, requires that every state classify its water bodies according to their degree of fertility and for those that are determined to be excessively fertile, develop nutrient control programs in order to reduce the excessive fertility. One of the initial basic questions which must be answered is whether water bodies of the southern US behave in a similar way to water bodies in the northern US with respect to their nutrient load-eutrophication response relationships. It is conceivable that the longer growing seasons, the warmer waters in both summer and winter, and the lack of any significant ice cover during the winter could cause lakes and impoundments in the southern part of the US to utilize their nutrient inputs to a different degree than northern lakes and, thereby, limit the applicability of the studies conducted on northern water bodies as a basis for the development of eutrophication control programs.

In an effort to provide information on nutrient load-response relationships, Dr. Lee contacted the Dallas Water Utilities (DWU), the Texas Water Quality Board, the US EPA Eutrophication Program, and others about the possibility of funding this study. It was arranged to support the study by funds from the North Central

Texas Council of Governments "208" planning money which was derived from the US EPA. Further, substantial funding for this study was derived from the Dallas Water Utilities, the University of Texas at Dallas, and EnviroQual Consultants & Laboratories of Fort Collins, Colorado. The total funding level for this study was approximately $300,000 over a 15-month period, with about half the funds being derived from the NCTCOG "208" money. While the funding achieved was considerably less than that needed to conduct the study as originally envisioned by Dr. Lee, it was adequate to develop sufficient information to answer one of the primary questions of this investigation, namely, do water bodies in the southern US behave the same with respect to nutrient load-response relationships as water bodies in other parts of the world, especially those in north temperate climates for which there is a considerable amount of Literature on eutrophication-response relationships?

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ABSTRACT

A 15-month study of the nutrient (nitrogen and phosphorus) inputs and limnological characteristics of Lake Ray Hubbard, an impoundment located near Dallas, Texas, has shown that planktonic algal numbers are primarily controlled by the input of available phosphorus to the water body. The phosphorus inputs are to arms of this lake which show varying degrees of eutrophication related to the overall phosphorus load. The excessive fertilization of the parts of Lake Ray Hubbard cause the Rowlett Arm of the lake (most heavily phosphorus and nitrogen loaded) to have characteristics typical of hypereutrophic lakes. The Main Body of the lake has a considerably reduced phosphorus load compared to the arms as a result of phosphorus removal within the arms of the lake.

The excessive fertilization of Lake Ray Hubbard results in a significant impairment of the use of the water in the Main Body of the lake for domestic water supply purposes because of algal-related tastes and odors in treated waters derived from this source. The Rowlett Arm of the lake also has significantly impaired water quality with respect to recreational use of this area.

The primary source of phosphorus for Lake Ray Hubbard is the discharge of this element in the domestic wastewater effluents from several municipalities located in the watershed. Forty-six percent of the total phosphorus input to Lake Ray Hubbard is derived from domestic wastewater effluents. Phosphorus has been found to be the key limiting element controlling excessive algal growths within this lake.

The initiation of advanced waste treatment practices involving 80 to 90 percent phosphorus removal can reduce the total phosphorus load to Lake Ray Hubbard to 61 percent of the current total load.

Failure to initiate phosphorus control programs in domestic wastewaters by the year 1990 will result in a 45 percent increase in total phosphorus load to the lake. Initiation of advanced waste treatment for phosphorus removal at this time and its continued practice for all domestic wastewaters added to the lake will cause the 1990 total phosphorus load to Lake Ray Hubbard to be approximately 67 percent of the current phosphorus load.

The Lake I deterministic model, originally developed for Lake Ontario, was found to not be applicable to Lake Ray Hubbard without substantial modifications. The OECD Eutrophication model for predicting the average chlorophyll a concentration in the lake from the phosphorus load to the lake was found to be applicable to Lake Ray Hubbard. During the study period, the average chlorophyll a concentration was 11.5 μg/l. The OECD model predicts that this could be reduced to 7.5 μg/l with 85 percent removal of phosphorus from domestic wastewaters due to advanced wastewater treatment. The model also points out the importance of the Lake Lavon flows on influencing water quality in the Main Body of Lake Ray Hubbard by predicting an average chlorophyll a concentration of 6.5 μg/l during the study if water had been released from Lake Lavon to the East Fork of the Trinity River. The importance of the flow from Lake Lavon and the other tributaries on influencing the chlorophyll a concentration in Lake Ray Hubbard is demonstrated by the impact of the ten-year high and ten-year low flow. Using the 1976-1977 nutrient loads, the predicted ten-year high flow chlorophyll a concentration in the Main Body of Lake Ray Hubbard is 10 μg/l, and the predicted low flow concentration is 16 μg/l.

EXECUTIVE SUMMARY

A large number of impoundments have been constructed in northeast Texas to minimize the impact of extended drought periods on domestic water supplies. Cost-effective water quality management in these impoundments requires an understanding of the impact of various types of contaminants added to the impoundments from the watersheds on water quality. One of the areas of particular concern for water quality management in these impoundments is the problem of excessive growth of algae and other aquatic plants. Prior to the completion of this study, there was essentially no information on aquatic plant nutrient load-impoundment water quality response relationships for these water bodies. A 15-month study (May 1976 to August 1977) was initiated at the University of Texas at Dallas (UTD) to examine nutrient load-eutrophication response relationships for Lake Ray Hubbard. This impoundment is one of the water bodies in the Dallas-Ft. Worth region which is experiencing excessive fertilization problems which are causing impaired water use. The impairment is primarily manifested in tastes and odors in treated drinking water and reduced use of some parts of the lake for recreational purposes. This report presents the results of this study of nutrient load-response relationships for Lake Ray Hubbard.

The study was conducted at the University of Texas at Dallas with the assistance of the Dallas Water Utilities. The North Central Texas Council of Governments, US EPA, and the City of Dallas Water Utilities, provided financial support to UTD for this study. Further, the Dallas Water Utilities assisted in collection of samples, analysis of the samples and provided the flow data for the tributary streams. This report is authored by UTD staff and students. The data review, synthesis, conclusions, and recommendations are primarily the product of the efforts of G. Fred Lee, Elaine Meckel, Magda Abdul-Rahman, Mike Gary, William Cale, and Walter Rast at the University of Texas at Dallas.

In the north central Texas region, there is limited information on the amounts of aquatic plant nutrients entering the water bodies, the factors controlling algal growth in these waters, and the relationship between algal growth and water quality. The Lake Ray Hubbard study focused on determining the amounts of aquatic plant nutrients (primarily nitrogen and phosphorus compounds) entering the lake, the fate of these nutrients within the lake, and the response of the lake to the nutrients.

Lake Ray Hubbard is divided into four main areas by highway causeways and bridges; the Main Body, and three shallow arms. The hypolimnetic oxygen was depleted in the deepest part of the Main Body throughout the summer period of thermal stratification. The average total P concentration in the Main Body was generally near 0.05 mg P/1 and the soluble ortho P concentration was near 0.005 mg P/1. During periods of algal blooms, the concentrations were generally less than this amount and during low-growth periods, they were somewhat higher. The total N concentration was generally near 1.5 mg N/1 and the inorganic N concentration was near 0.1 mg N/1. The concentrations of N and P were significantly higher during the spring months due to runoff from the land. The uncorrected chlorophyll a concentration was generally near 10 μg/1 in the Main Body. The algal concentration was near 15,000 organisms/ml during the summer months when the blue-green algae were present in large numbers, and near 7,000 organisms/ml during the non-bloom periods of the winter months. During January, there was a bloom of diatoms with an average of 23,000 organisms/ml. Algal growth in the Main Body was limited by phosphorus throughout the study. The Main Body had the best water quality in the lake.

The average total P concentration in Rowlett Arm was generally near 0.25 mg P/l and the soluble ortho P concentration was near 0.03 mg P/l but there were large fluctuations in the concentrations from week to week. The total nitrogen concentration was near 2 mg N/l. The inorganic N concentration was less than 0.01 mg N/l during the summer months and near 1 mg N/l during the fall and spring months. The uncorrected chlorophyll a concentration was generally near 50 µg/l in Rowlett Arm. The algal concentration was near 15,000 organisms/ml but large blooms of diatoms caused the number to increase to 45,000 organisms/ml in November and 60,000 organisms/ml in January. Algal growth in Rowlett Arm was most likely limited by nitrogen during the summer and phosphorus during the winter months; however, reduced light penetration due to turbidity may have been limiting algal growth to a certain extent in this arm of the lake. The water quality of Rowlett Arm is significantly degraded and recreational uses of the lake are impaired.

The water quality of Muddy Creek Arm and the East Fork Arm is intermediate between the Main Body and Rowlett Arm as measured by chlorophyll a concentration and algal numbers. The nitrogen and phosphorus concentrations indicate that phosphorus is most likely controlling algal growth in these arms of the lake.

The problems of obnoxious tastes and odors apparently caused by excessive fertilization of the lake have been sufficiently severe so that the City of Dallas Water Utilities does not use this lake as a raw water supply during certain times of the year. There appears to be no relationship between algal numbers and types in the water and tastes and odors in the treated water derived from the lake.

The annual loads of nitrogen and phosphorus to Lake Ray Hubbard were determined in this study. It was found that 46 percent of the total annual loading of 247,000 kg P/year was derived from the sewage treatment plants which discharge into tributaries of Lake Ray Hubbard; approximately 4 percent came from the atmosphere and the remaining 50 percent came from land runoff. Over 76 percent of the total P load enters Lake Ray Hubbard from Rowlett Creek which receives the effluents from the Garland and Plano sewage treatment plants. This study has shown that over 95 percent of the phosphorus entering the arms of the lake is removed before it reaches the Main Body.

The Lake I deterministic model, originally developed for Lake Ontario, was found to not be applicable to Lake Ray Hubbard without substantial modifications. The US OECD eutrophication models developed by Vollenweider for predicting the trophic status and average chlorophyll a concentration, based on the phosphorus load and morphometric and hydrologic characteristics of the water body, and the model developed by Rast and Lee for predicting the average Secchi depth, based on these same characteristics, appear to be applicable to the Main Body and Rowlett Arm of Lake Ray Hubbard. These results indicate that Lake Ray Hubbard, a southern warm water body, shows the same nutrient load-eutrophication response relationship as north temperate water bodies.

The characteristics of the Main Body and Rowlett Arm, as a result of various hypothetical manipulations of the phosphorus load, including advanced waste treatment for phosphorus removal at the sewage treatment plants, reduction of phosphorus load by a phosphate detergent ban, and increased phosphorus load through increased population in the Lake Ray Hubbard watershed, were estimated. The effects of the normal flow in the East Fork of the Trinity River were also considered. Based on the 1976-77 phosphorus load, the Main Body and Rowlett Arm lie in the eutrophic zone of the Vollenweider phosphorus loading diagram. Relatively large changes in phosphorus load to the lake can occur and have limited impact on the positions of the Main Body and Rowlett Arm on the Vollenweider loading diagram. Even with 85 percent phosphorus removal from domestic wastewaters, the

Main Body would still be classified as eutrophic compared to other lakes, and Rowlett Arm would still be among some of the highest phosphorus loaded water bodies.

The average growing season epilimnetic chlorophyll a concentration in the Main Body was 11.5 µg/l during the study period. The US OECD model predicts that with 85 percent phosphorus removal from domestic wastewaters, the average summer chlorophyll a concentration would be reduced to 7.5 µg/l and with 35 percent removal (detergent phosphate ban), it would be reduced to 10 µg/l. It is concluded that a detergent phosphate ban would have little effect on water quality in the Main Body; whereas, 85 percent P removal from the domestic wastewaters entering tributaries of Lake Ray Hubbard would result in a significant improvement in the water quality of the Main Body in the future.

The study has shown that the flow from Lake Lavon, an upstream impoundment, has a significant effect on water quality in Lake Ray Hubbard. The hydraulic residence time of the Main Body of Lake Ray Hubbard, under normal flow conditions from Lake Lavon, would be significantly less than the hydraulic residence time during the study period. The decreased hydraulic residence time allows water quality to be maintained at the same level with a greater phosphorus load. The year to year variations in flow would have a pronounced effect on the water quality in the Main Body of Lake Ray Hubbard. The 1990 chlorophyll a concentration with normal Lake Lavon flow and no phosphorus removal from domestic wastewaters is predicted to be 7.5 µg/l. An average concentration of approximately 5 µg/l could be achieved in 1990 with 85 percent phosphorus removal at the sewage treatment plants.

The average Secchi depths of the Main Body and Rowlett Arm are lower than those predicted by the relationship between Secchi depth and normalized P load developed by Rast and Lee. The low Secchi depths are likely due to the suspended sediments in the water column. Since the volume and depth of the hypolimnion of Lake Ray Hubbard are small, the sediment oxygen demand is exerted at a greater rate than predicted by the model developed for the US OECD water bodies.

The results of this study strongly support the current approach of the US EPA of establishing phosphate management strategy based on the Vollenweider-US OECD Eutrophication models. Using this approach, it has been shown that the removal of 85 percent of the phosphorus in domestic wastewaters entering Lake Ray Hubbard will likely result in a small increase in water quality at this time. Unless removal of phosphorus from domestic wastewaters is initiated in the near future, there will be further deterioration in water quality in the lake associated with increased population within the lake's drainage basin. Initiation of 85 percent removal of phosphorus from domestic wastewaters will result in the 1990 water quality of the lake being somewhat better than the 1976-77 water quality.

This study has shown the need for additional research to relate nitrogen and phosphorus loads to a water body to the taste and odor characteristics of the finished drinking water derived from that water body. Until this type of relationship is available, it will be impossible to predict the impact a reduction in nutrient load will have on tastes and odors in the finished waters.

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SECTION 1

INTRODUCTION

North central Texas experiences periodic droughts. In order to minimize the impact of extended dry periods, the water utilities and the governmental agencies in this region have constructed a large number of impoundments. These impoundments, in addition to serving as a source of water for domestic water supplies, provide extremely valuable recreational areas for this part of the state. Public Law 92-500, the Amendments to the Federal Water Pollution Control Act, Section 314, requires that each state classify its water bodies relative to their productivity. For those water bodies that are determined to be excessively fertile, the state must, in accord with the provisions of this Act, develop a program for the control of excessive fertility.

As a result of the comparatively recent construction of many of the impoundments in north central Texas, there is limited information available on the amounts of aquatic plant nutrients entering the various water bodies in this region, the factors controlling algal growth in these waters, and the relationship between algal growth and water quality. As part of their National Eutrophication Survey, the US Environmental Protection Agency (US EPA) conducted limited studies on impoundments in north central Texas. Their reports indicate that these water bodies are classified as eutrophic. The basis of this classification is criteria largely developed for lakes in the eastern and midwestern US. Prior to the initiation of the current study, there were few studies devoted to factors controlling eutrophication in southern and southwestern water bodies. It is possible that water bodies in this region will utilize their nutrient loads somewhat differently than those of the upper Midwest. As discussed by Lee (1977), the magnitude of the parameters used to judge eutrophication-water quality relationships in north central Texas would likely be somewhat different than for lakes and impoundments in the upper Midwest. It is reasonable to conclude that the US EPA's classification of the impoundments in north central Texas as eutrophic, based on criteria largely developed in the upper Midwest, may not be appropriate. It is highly likely that the response of the public to a certain biomass of algae and/or macrophytes will be different in north central Texas than in Michigan or Wisconsin. It is clear that there is a need for research which would define nutrient load-response relationships for water bodies in this region. The Lake Ray Hubbard study was designed to provide this type of information. This effort represents a 15-month study conducted jointly by several of the University of Texas at Dallas (UTD) staff and graduate students and the Dallas Water Utilities (DWU). This study is part of the North Central Texas Council of Governments' (NCTCOG) "208" planning effort. This study will be of importance to the residents of this area in that it will define the expected benefits that may be derived from controlling aquatic plant nutrients to a certain degree in Lake Ray Hubbard. Even though the study is restricted to this lake, the results of the study will have applicability to other water bodies in this region when the general characteristics of those other water bodies are defined.

The Lake Ray Hubbard study focused on determining the amounts of aquatic plant nutrients (primarily nitrogen and phosphorus compounds) entering the lake, the fate of these nutrients within the lake, the factors limiting planktonic algal growth in the lake, and the response of the lake to the nutrients. In addition, information was obtained on the amounts of nutrients derived from various types of land use and the relationship between algal populations in Lake Ray Hubbard and taste and odor problems in the drinking water from the Dallas Water Utilities East Side Water Treatment Plant.

--- *Over the past 45 years, the Dallas Water Utilities Department has been actively involved in minimizing customer complaints regarding the odor or taste of the water. Powdered activated carbon treatment has been consistently used whenever raw water quality was such that taste or odor problems could be anticipated. An extensive taste and odor monitoring program was initiated many years ago in an effort to determine the activated carbon dosage necessary to produce a finished water which would please the customers. Data gathered over the years have shown that rapid changes in odor level or odor characteristic will always cause some customers to call and inquire about the water treatment process. It has also been demonstrated that most customers will accept a water with threshold odor level of five units or less if the odor characteristic does not change from the normal or if there is no accompanying disagreeable taste. However, if the odor characteristic is disagreeable or a detectable taste is involved, most customers will not accept water with threshold odor level of three units.

Taste and odor problems at the East Side Water Treatment Plant have been seasonal over the past 13 years of operation. During the summer months, the raw water odor usually increases and changes from a musty to a grassy or fishy characteristic. Planktonic studies were begun in 1965 to see if algal blooms could be correlated with the periods of taste and odor problems. This was indeed found to be the case. A bloom of green algae usually occurs in the early summer as the water temperature increases. This is followed by a bloom of blue-green algae which begins in late July or early August. Actinomycetes counts were begun several years ago, and these organisms were found to increase in number along with the blue-green algae.

Lake Ray Hubbard was completed and put into use as a water supply in 1972. Up to that time, the entire raw water supply

---

*This section prepared by F. Stone and J. Brown, DWU East Side Water Treatment Plant.

had been taken from Lake Tawakoni. Algal blooms seemed to have a greater effect on water quality in Lake Ray Hubbard than on water quality in Lake Tawakoni.

In 1974, the East Side Water Treatment Plant experienced its first extended taste and odor problem which could not be controlled by activated carbon treatment. Numerous customer complaints occurred as a result of the problem. During June, the water took on a fishy, grassy odor which seemed to get worse as the water aged in the distribution system. Several customers called each day to complain about their water. The problem improved for about two weeks and then got much worse. The water then took on the classic geosmin type odor (earthy, woody, swampy) during this period, and activated carbon treatment seemed to have a minimal effect. Many customer complaints were received each day throughout the remainder of the summer. The usual customer comment was that the water smelled and tasted "swampy." Many thought that untreated lake water was being pumped to their houses, and they were concerned about their safety. This points out one of the greatest concerns about taste and odor problems. Many people will turn away from a safe water supply which tastes unpalatable and turn to an unsafe supply which is palatable. The water odor returned to normal after the first hard rainfall in late August, and customer complaints ceased.

A similar problem occurred during the summers of 1975, 1976, and 1977; however, customer complaints were minimized by discontinuing the use of Lake Ray Hubbard water. During 1976, the use of water was discontinued from August 3 until September 23. The problem encountered during the summer of 1977 began July 25, when raw water with a "woody" odor characteristic was noticed. An attempt was made to blend the water with that from Tawakoni, but this was found to produce a finished water with a slight "woody" taste and was discontinued. On August 11, water from Lake Ray Hubbard was blended with Tawakoni water and found to produce an acceptable finished water. However, the next day, Lake Ray Hubbard water was found to be useable when it was prechlorinated to breakpoint. This oxidation step appeared to destroy the objectionable odor.

The water quality in Lake Ray Hubbard during the summer of 1977 was similar to that of the past several summers and should be considered typical. It should also be noted that many taste and odor complaints were received from the areas of the city receiving water from Lakes Lavon, Lewisville, and Grapevine. The finished water produced from these lakes had the same "woody," "muddy" taste as that from Lake Ray Hubbard in the past years. Super chlorination was tried at Bachman and Elm Fork Plants along with powdered activated carbon treatment, and this appeared to reduce the taste and odor to a level that was not detected by most customers.

The characteristics of the taste and odor in finished waters from all of the above noted lakes were similar, and the problems occurred at approximately the same time of year. Biological examinations showed blooms of blue-green algae in each lake during the time when the taste and odor was present. This lends further evidence to the connection between biological activity in the lakes and the taste and odor complaints received from customers using finished water from those lakes.

It is evident that excessive fertilization of Lake Ray Hubbard is causing significant water quality problems for DWU. Further, since the watershed is in an area of rapid urbanization, it is likely that increasing water quality problems will be encountered by the DWU as a result of increased nutrient loads from the watershed.

Recently, additional impetus has been provided for the control of excessive fertilization in water supply reservoirs such as Lake Ray Hubbard as a result of the finding that the trihalomethane content of a municipality's treated water was found to be related to the overall algal productivity of the raw water supply (Personal Communication, P. Foley, Ontario Ministry of the Environment, 1977). The trihalomethanes are low molecular-weight chlorinated hydrocarbons, exemplified by compounds such as chloroform. Compounds of this type are of concern in domestic water supplies because they are known carcinogens and may cause cancer in man. The US EPA is currently developing regulations for control of the trihalomethane content of domestic treated water. While this study did not examine the relationship between trihalomethane formation and the algal excretion of organics which lead to the production of these compounds upon chlorination, it is an area that deserves further study in order to determine if the results found in Canada are applicable to water supplies in north central Texas.

One of the primary goals of this study is the development of a nutrient load-lake response mathematical model that can describe the relationship between a given nitrogen and phosphorus load to Lake Ray Hubbard and the planktonic algal growth within the lake. Emphasis of this study was placed on understanding the relationship between the nutrient load of the chemical, such as nitrogen and phosphorus, which is likely to limit the total planktonic algal biomass in the lake, to the taste and odor problems experienced by DWU as a result of using Lake Ray Hubbard as a water supply source. The work in this area was limited to threshold odor numbers and characteristics of the odor of the water in Lake Ray Hubbard. In addition, measurements were made of algal numbers and dominant types and, beginning in December 1977, of the actinomycetes numbers.

The results of the Lake Ray Hubbard studies in this area will represent a valuable addition to the work that is currently being done by the American Water Works Association Committee on Quality Control in Reservoirs chaired by G. Fred Lee of the University of Texas at Dallas. This committee has essentially the same objectives as this study. It is focusing its attention on the effects of land use in a water supply reservoir's watershed and nutrient transport to the reservoir and the water quality problems associated with the aquatic plant nutrients.

SECTION 2

CONCLUSIONS

1. Lake Ray Hubbard is receiving excessive amounts of aquatic plant nutrients from its watershed. These nutrients are stimulating the growth of algae which results in impaired water quality, manifested primarily in tastes and odors in the domestic drinking water produced by the Dallas Water Utilities East Side Water Treatment Plant. There is also impairment of recreational use of some parts of the lake due to the excessive nutrient loading which stimulates algal growth.

2. Phosphorus is the key chemical element controlling algal growth in the Main Body of Lake Ray Hubbard.

3. The primary source of phosphorus to Lake Ray Hubbard is domestic wastewater treatment plants which discharge to tributaries of the lake from the cities of Garland, Plano, Wylie, and Rockwall. Approximately 60 percent of the phosphorus that enters the lake is derived from these sources.

4. The highway causeways and bridges divide Lake Ray Hubbard into several arms with restricted exchange of water between them, which results in each arm having a somewhat different water quality dependent on its nutrient input. Rowlett Arm, which receives the wastewater discharges from Plano and Garland, shows highly degraded water quality. Muddy Creek Arm has somewhat less degraded water quality than Rowlett Arm. It receives small amounts of municipal wastewater from the village of Wylie. The East Fork Arm has water quality similar to that of Muddy Creek Arm. It receives wastewaters from the Rockwall sewage treatment plant. The Main Body of the lake has the best overall water quality. The discharge of the wastewaters to tributaries which

enter through the arms of the lake results in appreciable phosphorus removal within the arm, thereby improving the overall water quality within the Main Body.

5. Eighty to ninety percent of the phosphorus being added to Lake Ray Hubbard from domestic wastewater sources can be readily removed by advanced waste treatment processes that are in routine use in the US, Canada, and various parts of Western Europe.

6. Eighty-five percent removal of the domestic wastewater phosphorus will result in a 40 percent reduction of the total phosphorus input to Lake Ray Hubbard and will result in some yet undefinable reduction in the taste and odor problems experienced by the Dallas Water Utilities in their attempts to use Lake Ray Hubbard as a raw water supply.

7. Phosphorus derived from the use of household cleaners typically contributes about 35 percent of the total phosphorus in domestic wastewaters. A ban on detergent phosphate in the Lake Ray Hubbard basin would result in approximately a 15 percent reduction in the total phosphorus load to Lake Ray Hubbard. This reduction would not likely result in any significant decrease in algal growth within the lake.

8. The Lake I model, developed for Lake Ontario, is not a suitable model to predict nutrient load-response relationships for Lake Ray Hubbard. It should not be used in its current form for estimating the benefits to be derived from altering the nutrient loads to Lake Ray Hubbard or other water bodies in this region. Before it can be used for Lake Ray Hubbard or other water bodies in this region, it must be modified, properly calibrated for each water body, and verified on several of the water bodies.

9. There is insufficient information available today on the other North Central Texas Council of Governments' impoundments to formulate cost-effective water quality management policies. Additional studies need to be done to define nutrient loads, limiting nutrients, and general nutrient chemistry within the impoundments.

10. The excessive fertilization of the water bodies in this region may be of human health significance to those who use this water as a domestic water supply due to the possibility of increased formation of trihalomethane compounds.

11. The relationship between algal numbers and types in the water and tastes and odors in the treated water derived from the lake is very poorly understood. No direct correlation could be found from this study which would be indicative of a nutrient load-taste and odor characteristics of finished water relationship for Lake Ray Hubbard. Until this type of relationship is available, it will be impossible to predict, with any degree of reliability, the impact a reduction in nutrient load will have on tastes and odors in the finished waters. This relationship is likely dependent on a number of factors including specific types of organisms, the age of the organisms, and the method used in water treatment.

12. The OECD Eutrophication Study modeling approach has been found to accurately predict the chlorophyll level in the Main Body of Lake Ray Hubbard based on the phosphorus load to this part of the lake. This finding is of major importance in that it shows that this water body and quite likely other water bodies in this part of the US have phosphorus load-chlorophyll response relationships similar to those in water bodies in north temperate latitudes. This finding is also of importance in that it allows estimates to be made of the changes in water quality of Lake Ray Hubbard as a function of altering the phosphorus load to this lake.

13. The annual average chlorophyll concentration in the Main Body of Lake Ray Hubbard was approximately 11.5 µg/l during the study period. Under normal flow conditions and current phosphorus load, the expected annual average chlorophyll concentration would be approximately 6.5 µg/l in the Main Body. Under normal flow conditions and if the expected increase in population in the Lake Ray Hubbard watershed occurs, by 1990, the annual average chlorophyll concentration will increase to

approximately 7.5 μg/l in the Main Body. With 85 percent removal of phosphorus from domestic wastewaters discharged into Lake Ray Hubbard tributaries, and under normal flow conditions, by 1990, the annual average chlorophyll concentration in the Main Body will be on the order of 5 μg/l. It is concluded that controlling the phosphorus input to Lake Ray Hubbard through advanced wastewater treatment processes could have a significant impact on the overall water quality in the Rowlett Arm and, to a lesser extent, in the Main Body of the lake.

SECTION 3

RECOMMENDATIONS

1. It is recommended that the North Central Texas Council of Governments immediately work toward implementation of at least 85 percent phosphorus removal from the domestic wastewater treatment plants of Plano, Garland, and Rockwall. With respect to Garland, this should apply only to the Garland-Rowlett treatment plant at this time. The implementation of this recommendation with respect to Plano and Garland would not likely require extensive modification of the "new" wastewater treatment plants; it would require that these treatment plants be operated to provide phosphorus removal as well as removal of BOD and suspended solids.

2. It is recommended that the North Central Texas Council of Governments utilize the OECD-Eutrophication nutrient load-eutrophication response relationships for estimating the benefits to be derived from altering the nutrient inputs to impoundments in this region.

3. It is recommended that a monitoring program be established on each of the impoundments within the NCTCOG area of jurisdiction which would provide a data base to judge the relationship between the nutrient load and water quality within the water bodies.

4. It is recommended that studies be conducted to determine the effect, if any, excessive fertilization of the water bodies in the NCTCOG area has on the formation of trihalomethane compounds in treated waters derived from these water bodies.

5. It is recommended that studies be initiated which are specifically directed toward examining the relationship between organic compounds excreted by algae and other organisms and the taste and odor characteristics of the finished water.

6. It is recommended that nutrient control programs be implemented in the NCTCOG area to control total numbers of algae as measured by chlorophyll, until such time as information is available which would allow formulation of relationships between nutrient load and taste and odor intensity characteristics in finished drinking water.

7. It is recommended that the Dallas Water Utilities continue the Lake Ray Hubbard study in an abbreviated form for a period of no less than two years in order to determine the magnitude of the year-to-year variability on the lake's nutrient load-response relationship. This additional study is also needed to evaluate the impact of normal flow from Lake Lavon to Lake Ray Hubbard and the City of Plano's alteration of its wastewater treatment methods during the study period.

SECTION 4

CHARACTERISTICS OF LAKE RAY HUBBARD AND ITS WATERSHED*

Lake Ray Hubbard is located on the East Fork of the Trinity River in north central Texas (Figure 1), approximately 16 km northeast of Dallas as shown in Figure 2. The East Fork of the Trinity River rises in southern Grayson County in north central Texas and flows in a general southerly direction to its junction with the Trinity River about 134 km above its mouth. The East Fork drains an area of 3,390 km^2, which includes parts of Grayson, Fannin, Hunt, Collin, Dallas, Rockwall, and Kaufman Counties (see Figure 2). The watershed has a length of about 84 km along the major axis of the valley and a maximum width of about 48 km. The watershed of the East Fork of the Trinity River generally has wide fertile alluvial valleys consisting of Trinity and Catalpa clays. With the exception of the alluvial valleys and the Coastal Prairie soils near the mouth, the East Fork watershed lies almost entirely within the fertile Blackland Prairie soil region. These soils are highly productive but are subject to erosion when unprotected. Because of the areas of deep rich soils and the favorable climate, the greater part of the land in this watershed is devoted to extensive cultivation with grazing being practiced only on the steepest slopes and wooded bottom lands.

The Lake Ray Hubbard dam site lies in the northeastern part of Dallas County and the northwestern part of Kaufman County

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*Portions of this section were taken from a report to DWU (1960) by Forrest & Cotton Engineers.

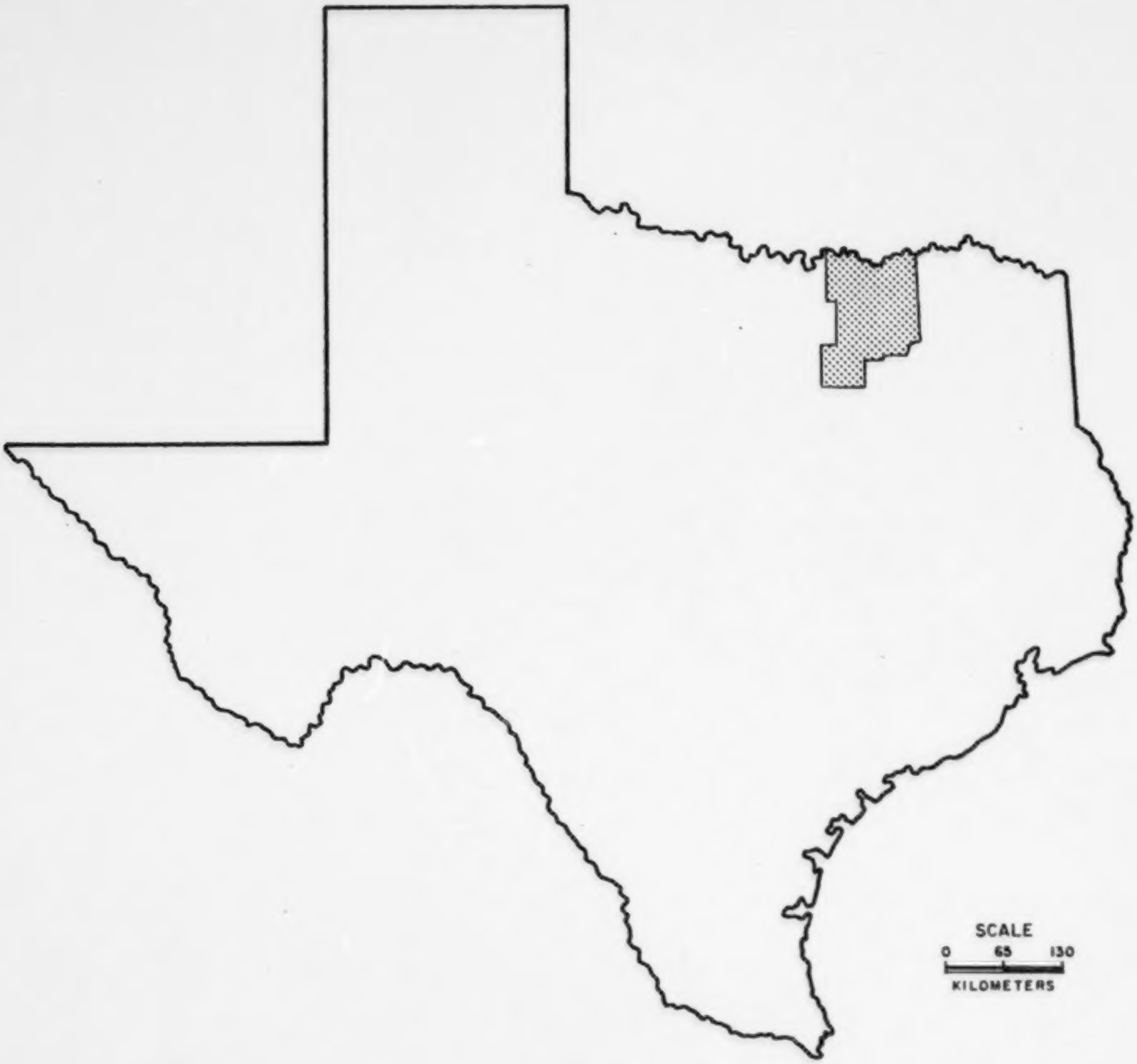
SCALE
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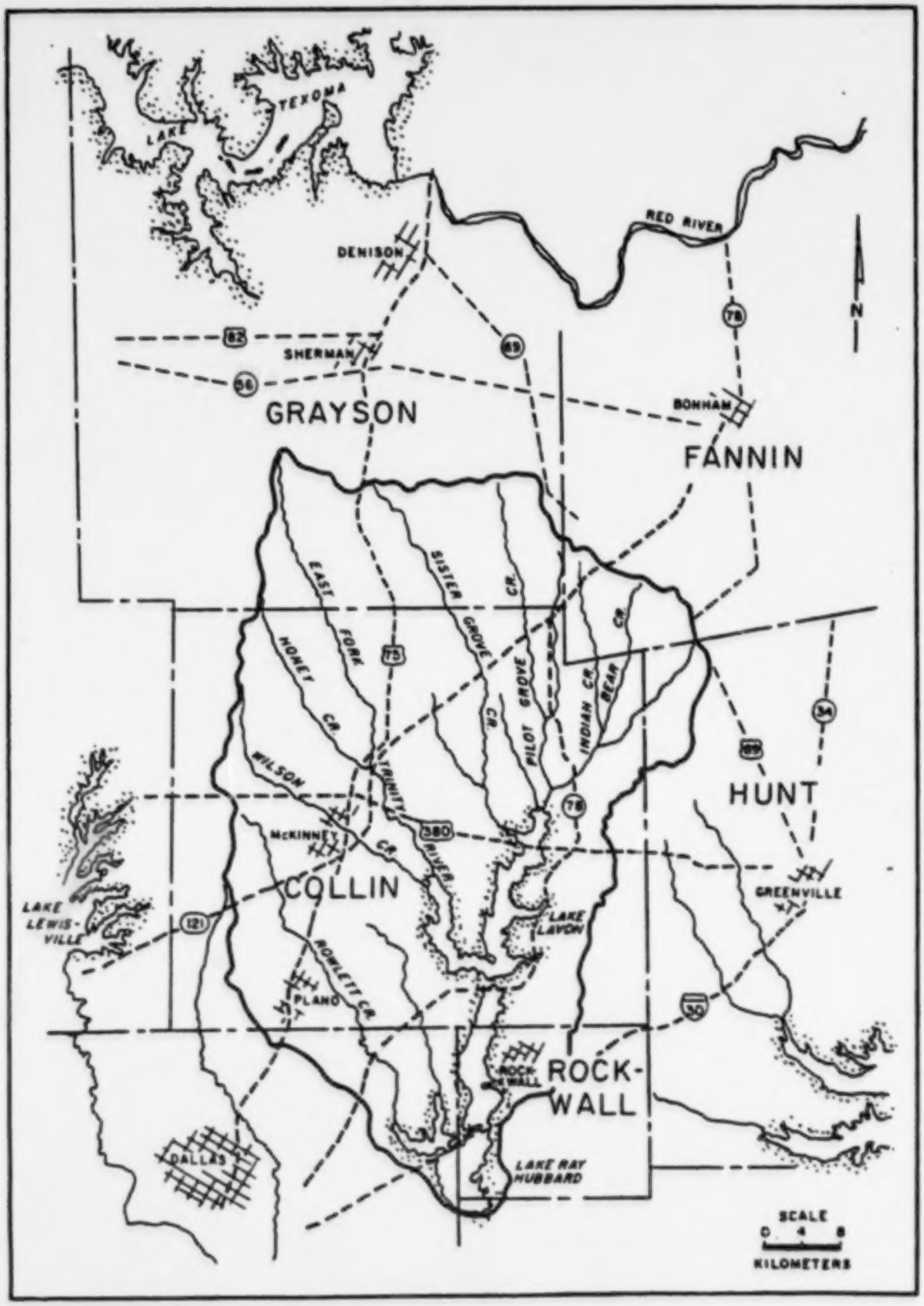
Figure 1. Location of study area in Texas


LAKE TEXOMA
RED RIVER
DENISON
82
SHERMAN
56
69
78
N
BONHAM
GRAYSON
FANNIN
EAST FORK
SISTER GROVE CR.
CR.
HONEY CR.
75
PILOT GROVE CR.
INDIAN CR.
BEAR CR.
34
69
HUNT
WILSON CR.
TRINITY RIVER
380
78
McKINNEY
COLLIN
LAKE LAVON
GREENVILLE
LAKE LEWISVILLE
121
PLANO
ROWLETT CR.
30
ROCK-WALL
ROCKWALL
DALLAS
LAKE RAY HUBBARD
SCALE
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KILOMETERS

Figure 2. Location of Lake Ray Hubbard

at river mile 31.8 on the East Fork of the Trinity River, 2.9 km upstream from US Highway 80 at the East Fork crossing. The dam controls runoff from 2,774 km^2, of which 28 percent is uncontrolled, and 72 percent is partially controlled by the Lake Lavon dam. The 780 km^2 unregulated portions of the watershed extend into portions of northeast Dallas, northwest Kaufman, southwest Rockwall, and southeast Collin counties.

LAND USE IN THE LAKE RAY HUBBARD WATERSHED

The amounts and forms of nitrogen and phosphorus entering a lake are significantly affected by the land use within the watershed. In order to predict the potential impact of further urbanization of the Lake Ray Hubbard watershed on water quality in the lake, it is necessary to ascertain the amounts of nutrients (nitrogen and phosphorus) derived from various types of land use. One of the areas investigated in this study was the relationship between the amounts of nutrients transported in a stream to the land use within the stream's watershed. Land use within the Lake Ray Hubbard watershed was divided into categories by NCTCOG. DWU aggregated the NCTCOG data into six major land use types: urban, cropland, grassland, forest, water, and a category which includes all other types of land use. Table 1 presents the area and percent of land in each land use category for the watersheds of each of the tributaries.

The total area of the Lake Ray Hubbard watershed including the East Fork of the Trinity River below Lake Lavon is 780 km^2. A major part of the watershed is rural with cropland comprising 34.2 percent of the total and 36 percent being grassland. Urban areas comprise 13.5 percent, 3.9 percent of the watershed is forested, and 11.8 percent is occupied by water.

During the study period, Rowlett Creek was the major tributary and source of nutrients to Lake Ray Hubbard. The upper Rowlett Creek watershed, including Spring Creek, has an area of 312 km^2. A large proportion of the watershed is rural with 45.1 percent being cropland and 33.5 percent being grassland. The

TABLE 1. LAND USE IN LAKE RAY HUBBARD WATERSHEDS*

Watershed and Sampling Station Number		Urban		Rural				Forest		Water		Other**		Total	
				Crop		Grass									
		km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%
Upper Rowlett Creek + Spring Creek	(15)	50.2	16.1	141	45.1	104	33.5	16.0	5.1	0	0	0.66	0.2	312	100
Lower Rowlett Creek		17.4	38.9	12.4	27.6	11.9	26.6	3.05	6.8	0	0	0.04	0.1	44.8	100
Muddy Creek	(16)	8.53	8.3	48.5	47.2	42.2	41.0	2.57	2.5	0.82	0.8	0.21	0.2	103	100
Cottonwood Creek	(17)	1.12	13.0	3.60	41.6	3.37	38.9	0.56	6.5	0	0	0	0	8.65	100
Rush Creek	(18)	1.90	15.8	1.78	14.8	8.03	66.9	0.06	0.5	0.08	0.7	0.16	1.3	12.0	100
Camp Creek	(20)	2.31	3.0	23.6	30.7	46.5	60.5	3.61	4.7	0.69	0.9	0.46	0.2	76.9	100
Squabble Creek	(21)	2.53	17.6	1.84	12.8	8.96	62.2	0.17	1.2	0.04	3.0	0.46	3.2	14.4	100
Yankee Creek	(23)	0.76	16.9	0.33	7.4	3.17	70.6	0.23	5.1	0	0	0	0	4.49	100
Unmonitored Watershed (around East Fork, Rowlett and Muddy Creek Arms and Main Body)		20.2	17.6	34.0	29.6	53.4	46.4	4.32	3.8	0	0	2.99	2.6	115	100
Lake Surface		-	-	-	-	-	-	-	-	89.6	100	-	-	89.6	100
Total		105	13.5	267	34.2	281	36.0	30.6	3.9	91.6	11.8	4.67	0.6	780	100

Dash (-) indicates not applicable.

*NCTCOG data adjusted to correspond with DWU operations analysis figures for lake surface area. Based on lake surface elevation of 434.5 ft msl.

**Includes feedlots, marshes and strip mining.

urban areas occupy 16.1 percent of the total and 5.1 percent of the watershed is forested. Lower Rowlett Creek has a watershed of 44.8 km^2 which is predominantly rural. Cropland occupies 27.6 percent and grassland comprises 26.6 percent of the total watershed area. A significant amount (38.9) percent of the lower Rowlett Creek watershed is urban. The remaining 6.8 percent of the watershed is forested.

The smaller tributaries have approximately the same proportions of each of the major land use categories as the Rowlett Creek watershed; however, the watersheds are much smaller with the exception of the 103 km^2 Muddy Creek watershed. The watersheds of the other tributaries have areas ranging from 4.5 to 76.9 km^2. Grassland and cropland are predominant in the watersheds. The amount of grassland in the watersheds of the smaller tributaries ranges from 38.9 to 70.6 percent. Cropland is the second largest category of land use, ranging from 7.4 to 47.2 percent of the area of the watersheds. Urban areas comprise 3 to 17.6 percent, and forested areas range from 0.5 to 6.5 percent of the area of the watersheds.

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An independent assessment of the land use within the Lake Ray Hubbard watershed has been made by D. Cargo of the Environmental Sciences Program at UTD. This assessment involved the classification of land use in over 100 different categories which were field checked and annotated on aerial photographs.

The Lake Ray Hubbard watershed extends over an area of about 82,700 hectares, which is about 817 square kilometers. About 75 percent (617 km^2) of the total area of the watershed is considered rural, while urban areas occupy about 14 percent (109 km^2) and water 11 percent (92 km^2). These figures are indicative of the watershed in the summer and fall of 1976 and are presented in further detail in Table 2.

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*This section was authored by D. Cargo and M. Nungesser.

TABLE 2. LAKE RAY HUBBARD WATERSHED LAND USES, FALL 1976

Type	Hectares	Square Kilometers	Percent of Total
All Urban	11,013	108.8	13.3
Urban	7,384	73.0	8.9
Highways, roads	3,629	35.8	4.4
Water	9,299	91.9	11.2
Rural	62,404	616.7	75.4
Pastures, grasslands, forests	39,298	388.4	47.5
Cropland	14,991	148.2	18.1
Other rural (buildings, open land)	8,115	80.1	9.8
TOTAL	82,716	817.4	100.0

Source: Douglas B. Cargo and Martha Nungesser. Environmental Land Use Characteristics of the Lake Ray Hubbard Watershed (A Survey), Environmental Sciences, The University of Texas at Dallas, 1976, unpublished.

Non-agricultural uses are classified as farm dwellings and farm storage areas such as barns, stables, and silos. Pastures, grasslands, and forests cover 388 km^2 or 47.5 percent of the land. Crops occupy 148 km^2, and all other rural uses (buildings and open land) occupy 80 km^2. Water occupies about 92 km^2. This is the areal measurement of Lake Ray Hubbard, plus all the creeks and tributaries, stock tanks and ponds.

Roads cover a large area of the watershed. The 36 km^2 of roads and their rights-of-way are equal to almost half of the urban land in the watershed. This acreage includes the residential streets and alleys, urban collectors, state highways, farm-to-market roads, US highways and expressways, and the private rural roads between pastures. The urbanized surface is only 8.9 percent of the watershed which is 73 km^2 but involves many types of land use, i.e., residential, manufacturing and commercial facilities, public buildings, and open space and utilities.

Soil Characteristics

A soil association is a landscape that has a distinctive proportional pattern of soils. It normally consists of one or more major soils and at least one minor soil, and is named for the major soil (US Department of Agriculture, 1969). The Lake Ray Hubbard watershed has five different soil associations: Houston Black-Austin-Dalco Association; Houston Association; Trinity Association; Houston Black-Burleson Association; and Ferris-Heiden Association. According to the Collin and Dallas County Soil Surveys, these soil associations have the following characteristics: (US Department of Agriculture, 1975).

1. Houston Black-Austin-Dalco Association - gently sloping, well-drained calcareous clayey soils. They are deep soils with dark grayish, black surfaces and are developed on uplands and underlain with chalk.
2. Houston Association - gently sloping, moderately well to well-drained calcareous soils. They have a thick,

dark gray brown surface and are underlain at about 50 inches by chalk or marl.

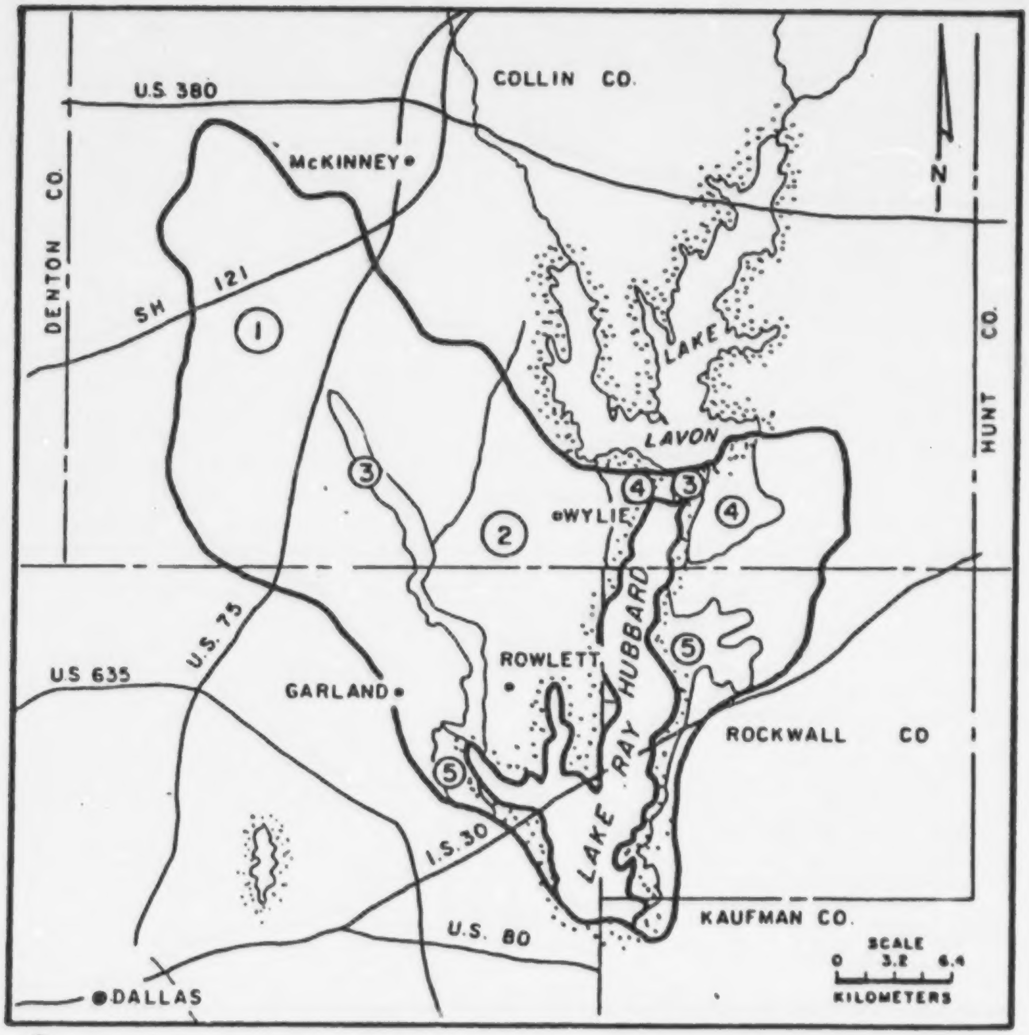
3. Trinity Association - nearly level bottomland soils, poorly drained calcareous clayey soils of flooded plains. They have a dark gray clay surface and are underlain at 6 to 20 feet with sand and gravel.
4. Houston Black-Burleson Association - These are nearly level to gently-sloping soils on stream terraces. They are deep, dark gray-brown calcareous clayey soils.
5. Ferris Heiden Association - steep slopes, excessively-drained calcareous soils over shales and marl.

The areal locations of each of these soil associations within the Lake Ray Hubbard watershed are shown in Figure 3.

Agricultural Practices

The agricultural practices which are carried on in the watershed are varied. Basically, these practices are of two main categories: crop farming and animal husbandry. Crop farming in the watershed primarily involves cotton, wheat, and grain sorghum. These crops are usually rotated per field so as not to deplete the soil of its nutrients. Because of this rotation, a survey at only one point in time will not give accurate figures for the three different types of crops. If, however, the majority of these crops are rotated in the same field, then the survey may be accurate. However, one of the management techniques that area farmers use is to allow some of their fields to return to grass or hay as part of regular crop rotation schemes. In these instances, the total cropland figures may be in error as the land use survey recorded the land use at the time of the survey and not some point in the past or present. With these constraints, Table 2 indicates that total cropland within the watershed was about 148 km^2 or 18 percent of the total watershed.

Animal husbandry in the Lake Ray Hubbard watershed is primarily focused on raising animal stock, which includes beef and


U.S. 380
COLLIN CO.
McKINNEY
N
DENTON CO.
SH 121
①
LAKE
LAVON
HUNT CO.
③
②
WYLIE
④
③
④
U.S. 75
⑤
ROWLETT
US 635
GARLAND
LAKE RAY HUBBARD
ROCKWALL CO
⑤
I.S. 30
KAUFMAN CO.
U.S. 80
SCALE
0 3.2 6.4
KILOMETERS
DALLAS

① Houston Black - Austin Dalco, gentle slope over chalk

② Houston, level to sloping clayey soils

③ Trinity, level bottomland soils, poorly drained

④ Houston Black - Burleson, level to gentle slope, deep, clayey soils stream terraces

⑤ Ferris-Heiden, sloping to steep slope clayey soils

Figure 3 . Soil associations in the Lake Ray Hubbard watershed horses. The kinds of land uses which are associated with this activity usually are grasslands and pastures.

Of the land-use categories within the watershed, pastures, grasslands, and forests occupy the largest area, 388 km^2, or about 47 percent of the total watershed area (see Table 2). While forests are included in this category, they represent a very small percentage of the total area.

Growth and Population Characteristics of the Lake Ray Hubbard Watershed

The watershed surrounding Lake Ray Hubbard has a variety of land uses. These land uses are not evenly mixed together to form a mosaic as one might expect in areas such as the great plains of Kansas. Rather, one or two land-use categories may occupy large segments of the watershed, while other land uses are mixed together to form completely different patterns in a much smaller portion of the watershed. The development of the watershed has not progressed uniformly at the same rate or with the same intensity. The results of these disparities have had and will continue to have dramatic effects upon fluvial dynamics of the watershed.

In general terms, the watershed has, within the past seven to eight years, experienced three kinds of growth (i.e., change in land use):

1. areas where little or no changes have occurred in land use;
2. areas where "filling-in" has occurred (i.e., the addition of more houses along streets, new subdivisions added to rural water system areas, ranchettes, etc.);
3. areas where "complete change" has taken place in land use. These changes for the most part are from rural to a total urban development. What were once cotton fields are now housing developments and subdivisions.

The areas of these different "growths" are near Rockwall, Sachse, Plano, Parker, Allen, and Garland (See Figure 4).

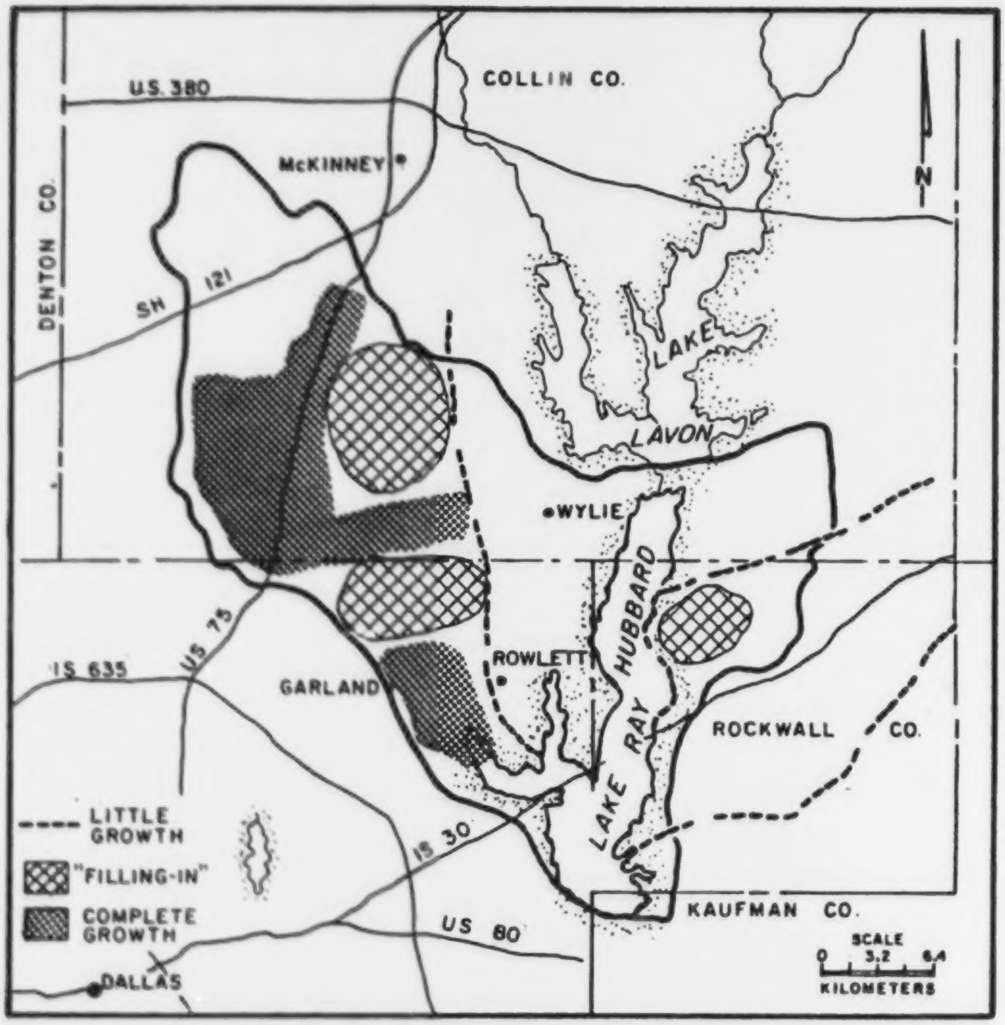

U.S. 380
COLLIN CO.
N
McKINNEY
DENTON CO.
SH 121
LAKE
LAVON
WYLIE
LAKE RAY HUBBARD
US 75
IS 635
GARLAND
ROWLETT
ROCKWALL CO.
LITTLE GROWTH
"FILLING-IN"
COMPLETE GROWTH
IS 30
US 80
KAUFMAN CO.
DALLAS
SCALE
0 3.2 6.4
KILOMETERS

Figure 4 . Areas of growth in the Lake Ray Hubbard watershed

For the most part, the lands immediately adjacent to Lake Ray Hubbard have not dramatically changed, i.e., from total rural to total urban. There has been some spotty development, but the character of the land-use mix has remained relatively unchanged. Indeed, these adjacent lands can be classified as areas where little or no appreciable change has taken place.

The greatest change in land use has taken place in western and northwestern portions of the watershed. These are analagous to the cities of Plano, Allen, Murphy, and Garland. Large areas of these cities have been transformed from cotton fields to housing subdivisions in a matter of a few years. This growth is now west of the watershed boundary in Plano, but Allen, to the north, is now experiencing rapid growth.

The "filling-in" process is occurring in places such as Parker, Sachse, and north Garland and Rockwall. Here the land has been owned for some time by developers who are now able to market these lands as pseudo ranches called "ranchettes." These "ranchettes" are usually two to five acres in size and can be used for keeping one or two horses.

In the other areas of the watershed, there is little change or growth occurring. The areas north of Plano and Allen, in and around Wylie, the Rowlett peninsula area, and east of Rockwall have experienced little change in comparison to the rapid expansion of Plano, but at the same time are not static or declining. All areas are growing in population, but the dynamics and speed with which this growth occurs varies.

Table 3 indicates the populations of the major communities within the Lake Ray Hubbard watershed. All have experienced growth; but the annual growth rate seems to be slowing. For example, between 1960 and 1970, Allen had a growth rate of 10.8 percent. During the years of 1970-1976, this decreased to 8.9 percent. This same general trend of a slight decrease in growth rate seems to be occurring for all communities within the Lake Ray Hubbard watershed except Rowlett which has seen an

TABLE 3. POPULATION OF THE MAJOR COMMUNITIES WITHIN THE LAKE RAY HUBBARD WATERSHED*

Community	1960	1970	1976	Change	
				1960-70	1970-76
Allen	659	1,940	3,300	1,281	1,360
Plano	3,695	17,872	44,150	14,177	26,278
Murphy	135	261	-	126	-
Sachse	359	777	-	412	-
Garland	38,501	81,437	123,250	42,936	41,813
Rowlett	1,015	2,243	4,100	1,228	1,857
Rockwall	2,166	3,121	4,200	955	1,079
Frisco	1,184	1,845	2,500	661	655
Wylie	1,804	2,675	2,750	871	75
Heath	175	449	-	274	-
Fate	191	329	-	138	-
Sunnyvale	969	995	-	26	-

Dash indicates data not available.

*Compiled from:

North Central Texas Council of Governments, 1970 Fact Book, Arlington, Texas, North Central Texas Council of Governments, 290 pp, 1971.

North Central Texas Council of Governments, Current Population Estimates, 1976, Arlington, Texas, North Central Texas Council of Governments, 1977.

increase in its growth rate. Rowlett had a 7.9 percent growth rate between the years of 1960 and 1970. For the time period of 1970 to 1976, it experienced a rate of growth in excess of 10 percent. This community, however, is the only one not corresponding to the overall general slow down in growth.

CHARACTERISTICS OF LAKE RAY HUBBARD

Morphological

The City of Dallas began construction of Lake Ray Hubbard in June 1964 as a water supply reservoir. The dam is a rolled earth-fill embankment with a total length of 3.8 km, including the spillway, which is located at the eastern end of the dam. The water supply intake structure for the City of Dallas, East Side Water Treatment Plant is located at the western end of the dam. In addition to serving as a water supply, Lake Ray Hubbard is a valuable recreational asset to the City of Dallas since it is used for fishing, swimming, and boating. The lake is also used by Dallas Power and Light Company for waste heat dissipation from a gas-fired electrical generating plant.

Lake Ray Hubbard has a volume of 5.5×10^8 m^3 at the normal pool elevation of 132.4 m (Table 4). The surface area of the lake is 8.8×10^7 m^2. The mean depth is 6.3 m and the maximum depth is 13 m. The total length of the lake is 23.3 km along the East Fork Arm. Highway and railroad bridges divide Lake Ray Hubbard into four main areas. Figure 5 is a contour map of Lake Ray Hubbard showing the arms and the Main Body. The area between the dam and the US Highway 67 bridges is referred to as the Main Body. This is the largest section with a volume of 3.5×10^8 m^3 and a surface area of 3.8×10^7 m^2. As shown in Figure 4, the East Fork Arm is the section north of the US Highway 67 bridge. This is a large shallow section of the lake (surface area = 3.6×10^7 m^2 and volume = 1.5×10^8 m^3), which is fed by several small tributaries including Squabble Creek which receives effluent from the Rockwall sewage treatment plant.

TABLE 4. LAKE RAY HUBBARD AREA AND VOLUME

Location	Surface Area (m^2)	Volume (m^3)
Main Body	3.8×10^7	3.5×10^8
East Fork Arm	3.6×10^7	1.5×10^8
Rowlett Arm	6.8×10^6	1.9×10^7
Muddy Creek Arm	7.5×10^6	3.0×10^7
Total	8.8×10^7	5.5×10^8

Lake Level 132.4 m

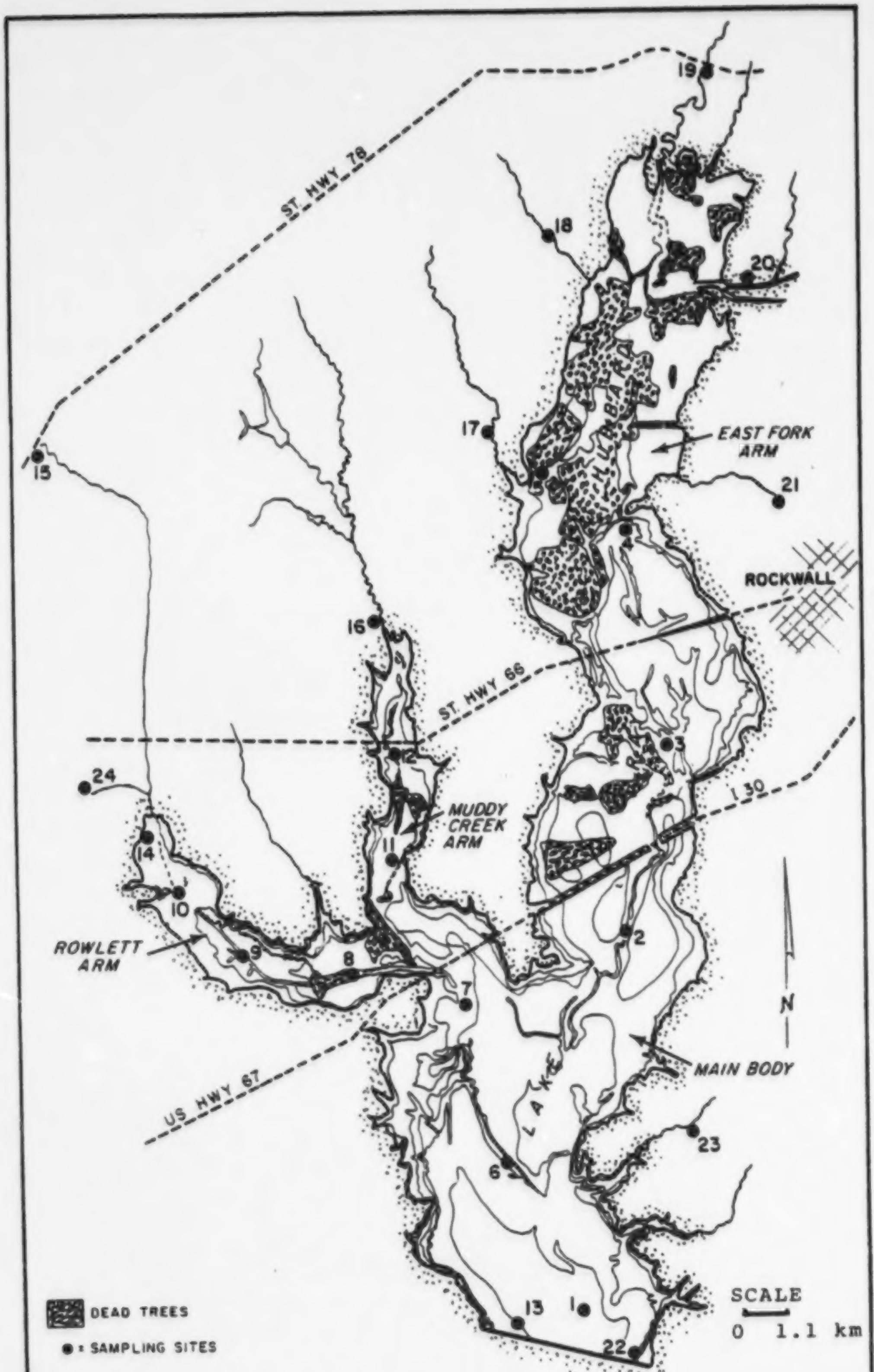

ST. HWY 78
19
18
20
HUBBARD
EAST FORK ARM
17
15
21
4
ROCKWALL
16
ST. HWY 66
3
12
24
MUDDY CREEK ARM
I 30
14
11
10
ROWLETT ARM
9
8
7
2
N
MAIN BODY
US HWY 67
LAKE
23
6
13
1
22
DEAD TREES
● = SAMPLING SITES
SCALE
0 1.1 km

Figure 5. Contour map of Lake Ray Hubbard

Under normal flow conditions, most of the water would enter Lake Ray Hubbard from the East Fork of the Trinity River, but due to the US Army Corps of Engineers' raising the normal pool elevation of Lake Lavon, essentially no water entered the lake from the East Fork during the study period. This makes the study period somewhat atypical hydrologically in that the hydraulic residence times of the East Fork Arm and the Main Body were longer than they would normally be with the flow from Lake Lavon. Ordinarily, the average flow from Lake Lavon would be 1.9×10^{11} l/year. If this flow had occurred during the study period, the hydraulic residence time of the Main Body of Lake Ray Hubbard would have been reduced from 3.5 to 1.2 years. The significance of this situation in influencing the results of this study are discussed in a subsequent section.

Rowlett Arm is the small area northwest of the US Highway 67 bridge which is fed by Rowlett Creek. This principal tributary receives treated domestic wastewaters from the cities of Garland and Plano. Rowlett Arm has a surface area of 6.8×10^6 m^2 and a volume of 1.9×10^7 m^3. The remaining small arm north of US Highway 67 is Muddy Creek Arm, fed by Muddy Creek which receives treated domestic wastewaters from Wylie. The surface area of Muddy Creek Arm is 7.5×10^6 m^2 and the volume is 3.0×10^7 m^3.

Climatological Characteristics of the Lake Ray Hubbard Area

The climate over the Lake Ray Hubbard watershed is generally mild with warm summers and cool winters. The periods of extreme cold that occur occasionally are short-lived, so that even in January, fair, mild weather occurs frequently. Likewise, the extremely high temperatures (above 38°C), that sometimes occur in summer, rarely last for an extended period. The prevailing surface winds are southerly. The mean annual temperature, mean monthly rainfall, and the mean monthly pan evaporation, as indicated by the 22 years of record at Lake Lavon, just north of Lake Ray Hubbard, are respectively, 18.3°C, 7.8 cm, and 17.4 cm. Table 5 presents some temperature and rainfall data for the Dallas area. The table shows that the average monthly high temperature

TABLE 5. CLIMATE OF DALLAS, TEXAS*

Month	Average Monthly High Temperature (°C)	Average Monthly Low Temperature (°C)	Average Rainfall (cm)	Mean No. Wet Days Per Month	Mean Humidity (%)
January	13	2	5.1	7	69
February	16	3	6.4	7	65
March	20	8	6.4	7	65
April	24	13	11.4	8	63
May	28	17	12.7	8	65
June	33	23	7.6	6	63
July	35	24	5.1	5	58
August	35	24	5.1	5	59
September	31	20	7.6	8	72
October	26	14	7.6	6	64
November	19	7	6.4	6	65
December	14	3	4.1	7	70

*Taken from the Dallas Morning News, Texas Almanac, 1974-1975.

ranges from 13°C in January to 35°C during July and August. The rainfall data show that the average monthly precipitation is fairly evenly distributed over the year with the most rain (11.4 to 12.7 cm) falling during April and May. The Dallas area is moderately humid with the mean humidity ranging from 58 to 72 percent.

Almost the entire "natural" flow into Lake Ray Hubbard is derived from surface runoff with seepage and springs contributing a minor portion of the flow. Due to the topography of the watershed, the character of the soils, and the nature of rainfall prevailing in the area, the stream flow varies widely with extended periods of very low flow and short periods of flows with high peaks. Figure 6 shows the average precipitation in the watershed and the flow in Rowlett Creek during the study period. As shown in the figure, the flow of Rowlett Creek was closely related to rainfall. During the dry periods, there was essentially no flow in the creek, whereas, during the heavy rainfalls, the creek flowed at rates up to 103,600 l/sec. During periods of low rainfall, the entire inflow to Lake Ray Hubbard consisted of effluent from the Plano, Garland, and several other North Texas Municipal Water District sewage treatment plants.

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The spring of 1976 was preceded by a very dry six-month period, September 1975 through February 1976. The effect of this dry period was to greatly reduce the soil moisture within the Lake Ray Hubbard watershed. Many of the rains in late 1976 were very light and, due to the dry watershed, produced no significant runoff.

From an overall point of view, precipitation in the Lake Ray Hubbard uncontrolled watershed (flow not coming from Lake Lavon) was near normal for the 15-month period, as judged by four rain gages located throughout the basin (see Table 6). Average watershed precipitation during the first twelve months (June 1976 through May 1977) was 90.2 cm. This is 6.6 cm lower

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*This section prepared by the Dallas Water Utilities.

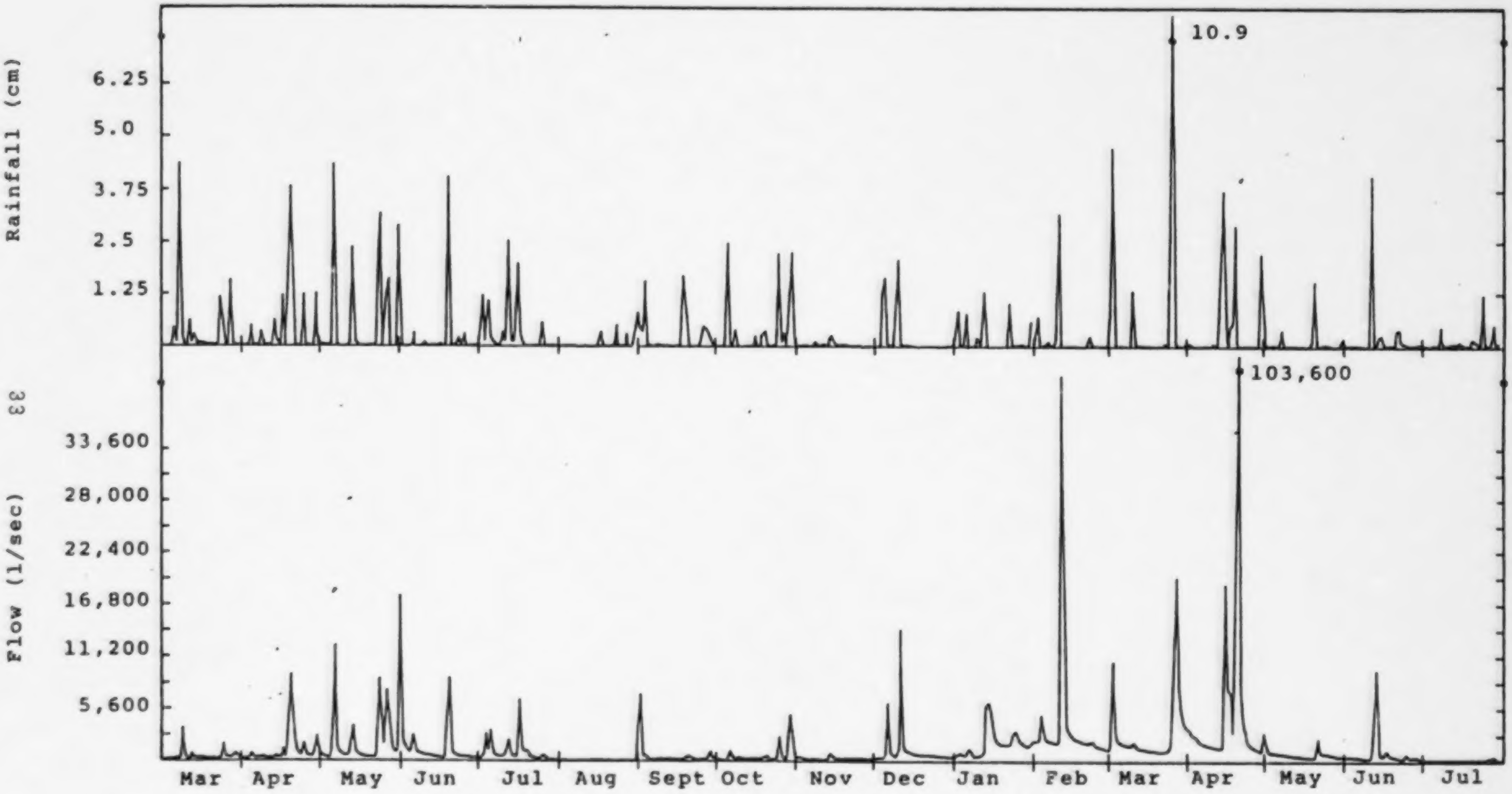

Figure 6. Precipitation - Lake Ray Hubbard and flow data - Rowlett Creek Hwy 78 Bridge

TABLE 6. PRECIPITATION DATA FOR THE LAKE RAY HUBBARD WATERSHED FOR THE STUDY PERIOD

	Precipitation (cm) Gage Location					
Month	Lavon Dam	McKinney	Richardson	Rockwall	Average of These Four Gages	Corresponding Month Period of Record Average at McKinney
6/76	5.59	5.77	5.44	2.90	4.93	9.35
7/76	15.93	9.02	9.86	9.65	11.13	6.45
8/76	2.31	2.39	1.30	0.56	1.65	5.54
9/76	6.55	11.38	6.50	8.08	8.13	10.13
10/76	9.25	10.19	8.08	11.53	9.75	7.65
11/76	0.91	1.52	1.45	0.94	1.22	6.71
12/76	7.98	7.19	6.73	7.19	7.26	5.46
1/77	6.78	8.20	6.40	7.21	7.16	5.46
2/77	6.12	6.35	3.45	7.01	5.74	6.63
3/77	19.51	17.22	12.50	24.18	18.36	8.03
4/77	13.06	13.74	10.16	10.59	11.89	12.24
5/77	2.74	3.45	2.34	3.68	3.05	13.13
6/77	6.17	5.00	0.81	7.04	4.75	-
7/77	7.21	0.30	0.81	4.50	3.20	-
8/77*	5.66	3.58	2.74	0.69	3.18	-
TOTAL					101.40	

Dash (-) indicates data not available.

*First 21 days of the month.

than the period of record annual average precipitation of 96.8 cm at McKinney located about 6.5 km north of the Lake Ray Hubbard watershed boundary. The summer 1977 rainfall was lower than normal with 11.2 cm being recorded in the watershed for June, July, and the first 21 days of August. Table 6 shows monthly precipitation at four locations, either within or near the uncontrolled watershed for the 15-month study period. For comparison, the average monthly precipitation over a 72-year period at McKinney is also listed. The summer 1977 rainfall is given for rain gages located within the Lake Ray Hubbard uncontrolled watershed.

The long period of record at McKinney shows that years with near average precipitation seldom occur. The average constitutes an average of highly variable numbers. Yearly totals commonly vary between 50 and 130 cm per year and are occasionally outside this range. Therefore, the first year of the study period experienced typical precipitation quite near normal. Also, comparison by month to average monthly rainfall is reasonably typical. The March 1977 rainfall, in excess of 18 cm, though not too unusual, would more typically have occurred in May. The May 1977 rainfall was as far below average as the March 1977 rainfall was above average. Seasonal precipitation was near the seasonal averages.

The summer 1977 rainfall was slightly more than half the summer average. Since summer rainfall is highly variable, summers with rainfall as low as, or lower than, the 1977 summer occur frequently.

ESTIMATES OF TRIBUTARY STREAM DISCHARGE DURING STUDY PERIOD ---*

Natural monthly inflows to Lake Ray Hubbard from the ungaged part of the watershed for the study period were estimated using a watershed runoff simulation model. Simulated flows were supported by stream discharge measurements made at sampling sites on many of the sampling days.

The watershed model - Hydrocomp Simulation Program written by Hycrocomp, Inc. of Palo Alto, California - simulates the hydrologic cycle. The processes in the hydrologic cycle are represented as a series of mathematical expressions. These processes include:

1. the interception of precipitation by vegetation (including parameters to represent the amount and type of vegetal cover),
2. the infiltration of precipitation which reaches the soil, and the direct runoff of precipitation falling on impervious area (including parameters to represent soil characteristics),
3. changes in soil moisture and subsequent effect on infiltration rate,
4. interflow of water horizontally through the soil and reemergence into the stream (also including parameters for soil characteristics),
5. overland flow (including parameters for surface characteristics),
6. evaporation and transpiration from ground storage, intercepted storage, and/or reservoir storage,
7. channel flow (with channel characteristic parameters).

When measurable watershed characteristics, such as drainage areas, surface slope, length of overland flow, vegetal cover, length and slope of stream channels, small reservoir volumes, etc., have been inputted, remaining parameters must be evaluated by calibration.

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*This section prepared by the Dallas Water Utilities.

The US Geological Survey (USGS) maintains two stream gages in the Lake Ray Hubbard watershed. One of these is located just downstream of the Lake Lavon discharge. The other is located on Rowlett Creek at Highway 78. The Rowlett Creek tributary flows from a 307 km^2 drainage area which has been gaged by a USGS streamflow gage since 1968. The Rowlett Creek gage was also used for model verification. The period of record from 1968 through 1973 was used for model calibration. Then, with final calibration parameter values, simulated Rowlett Creek flows after 1973 were compared to gaged flows. Simuated and gaged flows agreed within about 20 percent. The Rowlett Creek drainage is similar to other drainages in the Lake Ray Hubbard watershed. The differences which exist in impervious areas, average land slope, vegetal cover, and number of small reservoirs are recognized in the land segment (watershed area subdivision) parameters. The calibrated model was then used to estimate inflows to Lake Ray Hubbard from the remaining uncontrolled drainage area (463 km^2). Sewage treatment plant effluents were added to the simulated streamflows. During the study period, there were no inflows from Lake Lavon, the upstream reservoir.

Watershed simulation modeling is a reasonably reliable alternative to the more costly continuous streamflow measuring. Daily instantaneous flow measurements cannot be used alone for reliable estimates of flow. Intense rains frequently cause sharp rises and declines in stream flows; therefore, instantaneous flows cannot be used without knowing their positions on the inflow hydrographs.

The water budget method of reservoir inflow estimation has some merit for estimating inflows over a fairly long period of time. Reservoir level fluctuations due to wind complicate day-to-day estimates. Also, this method is sensitive to evaporation and transpiration.

Areal variation in rainfall is probably the major problem with watershed runoff simulation modeling. During the first part of the study period, as well as during the model calibration period, daily precipitation data were available only from gages at Rockwall, Richardson, McKinney, and the Lake Lavon Dam. Hourly precipitation was available only from the Lavon Dam station.

The watershed area was subdivided for modeling purposes into land segments. Daily precipitation was distributed to hours using the Lavon Dam hourly record, and these resulting rates were used for the watershed land segments. Four recording rain gages were installed and used after November 1976. Small discrepancies may occur between measured and simulated tributary flows due to very localized rainfall.

Estimated Flows of Each Tributary During the Study Period

The flows of Rowlett Creek were based on measurements by the USGS at the Highway 78 bridge (Station 15), and by the Garland sewage treatment plant at Station 24. The flows of the East Fork of the Trinity River at Highway 78 (Station 19) were measured by the USGS. The flows of all other tributaries were estimated using the Hydrocomp model. Table 7 presents the estimated annual flows between June 1976 and May 1977 at each tributary sampling station and at each of the sewage treatment plants which discharge into the tributaries. Examination of the table shows that Rowlett Creek was the major tributary during the study period with an annual flow of 6.4×10^{10} liters at Station 15. Muddy Creek and Camp Creek were the next largest tributaries with flows of 1.9×10^{10} l/yr and 1.2×10^{10} l/yr, respectively. All of the other tributaries were relatively small with flows ranging from 1.1×10^{7} l/yr in the East Fork of the Trinity River to 2×10^{9} l/yr in Squabble Creek. The low flow of the East Fork of the Trinity River was due to the raising of the water level of Lake Lavon by the Army Corps of Engineers. The normal flow of the East Fork of the Trinity

TABLE 7. ESTIMATED ANNUAL FLOWS OF TRIBUTARIES AND SEWAGE TREATMENT PLANTS

Sampling Station & Number		Annual Flows (liters)*
Rowlett Creek at Highway 78	(15)	6.4×10^{10}
Rowlett Creek at Miller Road	(14)	7.2×10^{10}
Muddy Creek at Liberty Grove Rd.	(16)	1.9×10^{10}
Cottonwood Creek at Stonewall Road	(17)	1.6×10^{9}
Rush Creek at Troy Road	(18)	1.6×10^{9}
East Fork of the Trinity River at Highway 78	(19)	1.1×10^{7}
Camp Creek at Highway 205	(20)	1.2×10^{10}
Squabble Creek at Highway 205	(21)	2.0×10^{9}
Yankee Creek at Meadowview Rd.	(23)	1.8×10^{8}
Garland Sewage Treatment Plant	(24)	5.6×10^{9}
Rockwall Sewage Treatment Plant		1.5×10^{6}
Plano Sewage Treatment Plant		4.8×10^{9}
Total Annual Flow**		1.1×10^{11}

*June 1976 to May 1977.

**Does not include flow at Station 14 or flow at the Plano Sewage Treatment Plant.

River is expected to be 1.9×10^{11} l/yr when Lake Lavon is at full pool. The total annual flow during the study period was 1.1×10^{11} l/yr which indicates that in normal years the East Fork flows will more than double the total annual flow to Lake Ray Hubbard.

Reliability of Estimated Nutrient Loads and Flows to Lake Ray Hubbard

In order to estimate the magnitude of the potential errors associated with the estimated flows, a comparison was made between the City of Dallas Water Utilities estimated flows at Station 15 (Rowlett Creek, Highway 78 bridge) and the USGS measured flows at this location. This was the location used by the Dallas Water Utilities for calibration of the Hydrocomp model for the Lake Ray Hubbard watershed. Therefore, this model should fit best for this station, and the errors are likely to be larger for the other sub-basins within the watershed.

Table 8 presents a comparison between the DWU Hydrocomp model estimates for Station 15 flows and those measured by the USGS for the period 5/76 to 7/77. From 5/76 through 10/76, only one rain gage was used to estimate precipitation which is the primary base for measuring flows. From 11/76 through 7/77, four rain gages were used for estimating flows.

The table shows that in some months as much as 85 percent difference existed between the Hydrocomp model estimated flows and those measured by the USGS. Typically, differences of 30 to 60 percent were encountered between the measured and estimated flows. The Hydrocomp model was tuned for estimating the annual flow so there could be significant differences between measured and estimated flows for shorter periods of time. With respect to the annual flows, the Hydrocomp model did a reasonably good job of estimating with only a 17 percent difference during the period when four rain gages were used, and a 20 percent difference during the period when one rain gage was in use. However, it did a relatively poor job on a monthly basis, especially in May 1977.

TABLE 8. A COMPARISON BETWEEN ESTIMATED AND MEASURED FLOWS FOR ROWLETT CREEK AT HIGHWAY 78

Date	Total Volume of Monthly Flow* Gaged ($m^3 \times 10^{-5}$)	Simulated ($m^3 \times 10^{-5}$)	Percent Difference
1 Rain Gage			
5/76	70.0	112.2	60
6/76	29.8	35.7	20
7/76	25.8	19.4	25
8/76	6.2	8.0	29
9/76	11.8	7.2	39
10/76	14.1	6.4	55
Total Volume for Period	157.7	188.9	20
4 Rain Gages			
11/76	6.3	2.5	61
12/76	33.3	11.8	65
1/77	50.8	19.1	62
2/77	90.2	43.6	52
3/77	170.6	226.4	33
4/77	185.2	126.9	31
5/77	17.6	32.7	85
6/77	21.9	14.5	34
7/77	5.0	6.9	32
Total Volume for Period	580.9	484.4	17

Drainage area = 307 square kilometers.

Difference between gaged and simulated (based on 4 rain gages) $96.5 \times 10^5\ m^3$ = 16.6 percent.

*Prepared by DWU.

Table 9 presents a comparison between the monthly total phosphorus loading at Station 15 using the mean monthly total P concentration and the estimated and gaged flows during the period when four rain gages were used for estimating precipitation. The monthly loadings, based on the Hydrocomp flow data, were as much as 20,000 kg/month low during February and 16,000 kg/month high during March.

As with the flow data, the difference between the loadings calculated from the measured flows and the loadings calculated from the estimated flows ranged from 31 to 85 percent. The difference of 20,000 kg/month is greater than the monthly loadings of all of the other tributaries and greater than the annual loadings of all tributaries except Muddy Creek.

The annual input of total phosphorus was determined to be 247,017 kg. If the atmospheric component and the loadings based on measured flows (Rockwall and Garland sewage treatment plant contributions, and the Station 15 loadings) are subtracted from the total loading, the remaining loading which was based on simulated flows is 47,691 kg P/yr. The overall difference between the loadings at Station 15, based on the estimated flows and the measured flows, is 17 percent. If it is assumed that the difference of 17 percent holds for the other tributaries, the total loading to the lake would change by 8,100 kg P/yr. Since the difference of 17 percent is based on the tributary for which the model was tuned, it is likely that the actual differences for the other tributaries would be larger. Even if the error was twice as large, the difference in the total loading to the lake would be 16,200 kg P/yr which is only 7 percent of the loading as measured in this study. These results point to the importance of accurately measured flows at the sewage treatment plants and Rowlett Creek. They also indicate that, even though on a short-term basis the Hydrocomp model used by the DWU does a relatively poor job of estimating flow, the magnitude of the errors for this study are insignificant since the flows for

TABLE 9. COMPARISON OF LOADINGS USING ACTUAL AND SIMULATED FLOWS

Date	Station 15 Avg Total P Concentration (mg/l)	Loading Gaged Flow (kg/month)	Loading Simulated Flow (kg/month)	Difference (kg/month)
11/76	0.813	513	203	310
12/76	1.275	4,256	1,505	2,751
1/77	0.934	4,750	1,782	2,968
2/77	4.263	38,497	18,615	19,882
3/77	2.842	48,556	64,408	-15,852
4/77	1.590	29,487	20,207	9,280
5/77	1.583	2,792	5,176	- 2,384
6/77	0.811	1,776	1,176	600
7/77	0.999	495	691	- 196
Total		131,122	113,763	17,359

which it was used were for tributaries which contributed a small part of the total annual phosphorus load. The flows associated with the important nutrient sources for the lake were either measured or estimated using other approaches. Caution should be exercised in using the Hydrocomp model for estimating flows of tributaries in the NCTCOG region where the tributary contributes a substantial part of the estimated nutrient load for the water body.

SECTION 5

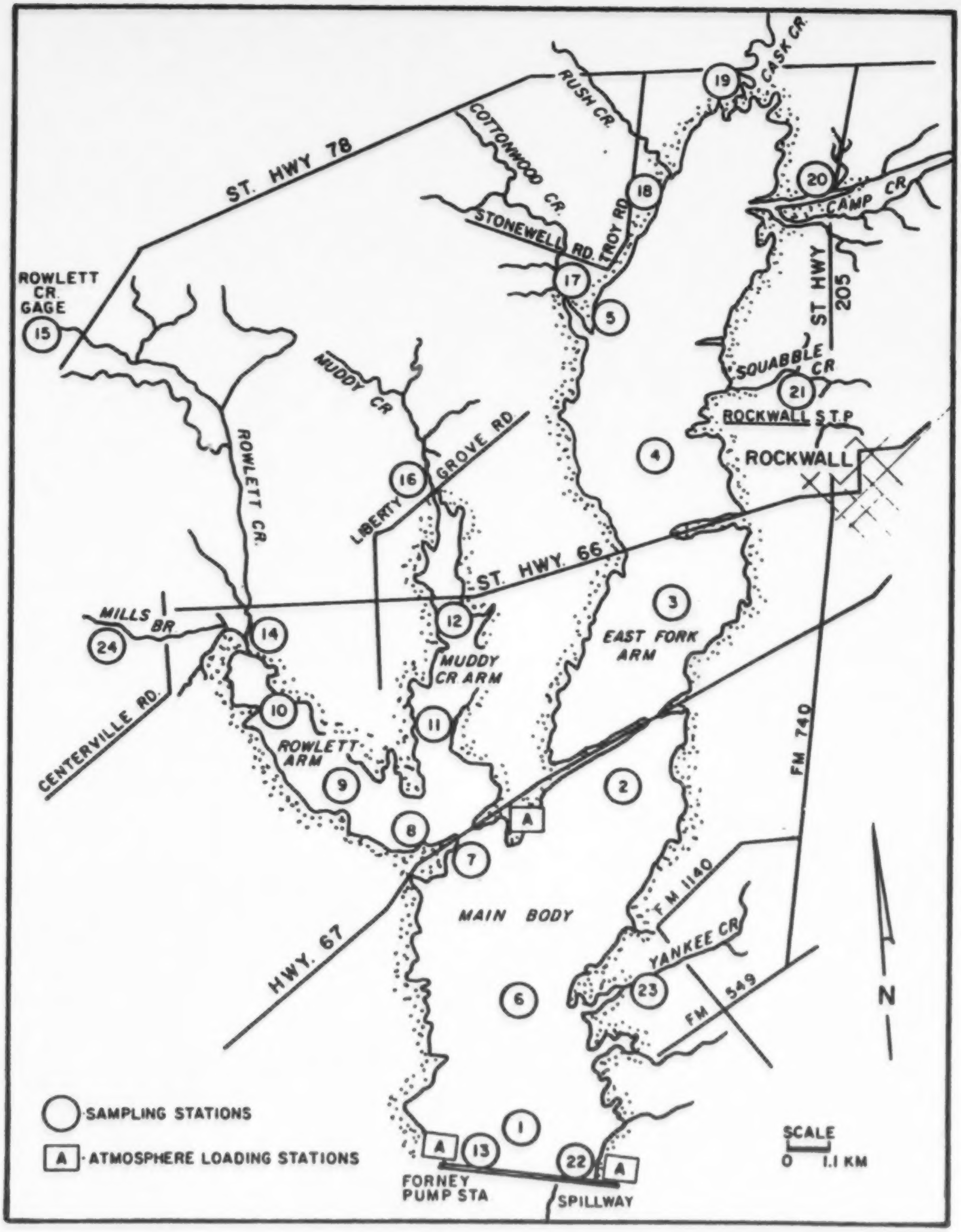
EXPERIMENTAL PROCEDURES

The analytical procedures and methods used in this study are discussed in this section.

SAMPLING METHODS AND LOCATIONS

Figure 7 is a map of the Lake Ray Hubbard area showing the lake, tributary, and atmospheric loading sampling stations. Several sampling stations were chosen in each of the four main areas of the lake in an attempt to characterize each area with respect to pertinent nutrient load-lake response parameters. Sampling stations were chosen on each of the tributaries to be above backwater areas of the lake, except for Stations 14 and 20. Station 14 is in the backwaters of Lake Ray Hubbard and, therefore, the concentrations of chemicals at this location are often influenced by the mixing of Lake Ray Hubbard water with Rowlett Creek water. This same problem existed at Station 20 on Camp Creek. This was the only location that was readily available for sampling this stream. The results of this station are influenced by the mixing of Lake Ray Hubbard water with Camp Creek water. The atmospheric loading sampling stations were located around the Main Body in an attempt to estimate the amounts of nutrients entering the lake from the atmosphere. Atmospheric loading stations were also located in Parker and Denison in order to determine the regional variability in loading from the atmosphere.

Twelve stations on Lake Ray Hubbard were sampled every week from June through August 1976. In September 1976, the


ST. HWY 78
RUSH CR.
CASK CR.
COTTONWOOD CR.
STONEWELL RD.
TROY RD
CAMP CR.
ST HWY 205
ROWLETT CR. GAGE
MUDDY CR
SQUABBLE CR
ROCKWALL S.T.P.
ROCKWALL
LIBERTY GROVE RD.
ROWLETT CR.
ST. HWY. 66
MILLS BR
EAST FORK ARM
MUDDY CR ARM
CENTERVILLE RD.
ROWLETT ARM
FM 740
A
FM 1140
MAIN BODY
HWY. 67
YANKEE CR
FM 549
N
SAMPLING STATIONS
A ATMOSPHERE LOADING STATIONS
FORNEY PUMP STA
SPILLWAY
SCALE
0 1.1 KM
1
2
3
4
5
6
7
8
9
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16
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19
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21
22
23
24

Figure 7. Sampling stations

number of stations was reduced to ten because of low water conditions that existed at Stations 5 and 10, and the sampling frequency was reduced to once every two weeks. The lake was sampled once a month from December 1976 to February 1977, every three weeks from March to May 1977, every two weeks during June and July 1977, and once a week during August 1977.

As shown in Figure 7, there were four sampling stations in the Main Body. There were originally four stations in Rowlett Arm, but the number was reduced to three after the summer of 1976. Muddy Creek Arm was sampled at two sites, and the East Fork Arm was sampled at three sites during the summer of 1976 and at two sites for the remainder of the study. Table 10 shows the total depth and the surface, mid, and bottom depths at which water samples were taken at each station.

The field work and sample collection on the lake were done from the DWU 19-foot Monarch boat tied, during the sampling, to a buoy marking each station. The water samples were usually collected with a Sears model number 340-26000 pump; however, a Van Dorn sampler was used on several occasions when the pump was not functional. Samples from Stations 12 and 14 were collected with the Van Dorn sampler from bridges. Samples for chemical analyses (pH, specific conductance, turbidity, total alkalinity, total hardness, suspended solids, chloride, ammonia, nitrate, nitrite, organic nitrogen, total phosphorus, soluble orthophosphate, chlorophyll a and pheophytin a) were collected at the surface (1 m) and bottom (0.5 m from the sediments) and at the mid-depth if the total depth was greater than 5 m. Algae and actinomycetes samples were collected at 1 m, and a threshold odor sample was collected at 2 m. A vertical tow for zooplankton was made with a 76 μ mesh plankton net at each station.

Dissolved oxygen (DO) and temperature measurements were made at 1 m intervals with a YSI Model 57 DO meter. Light penetration at 1 m intervals was measured with a Kahlsico Model 268 WA-310 submarine photometer. The Secchi depth was determined with a 20 cm Secchi disc. Beginning in March 1977,

TABLE 10. SAMPLING DEPTHS IN LAKE RAY HUBBARD

Station No.	Total Depth (m)	Sampling Depths (m) Surface	Mid	Bottom
1	13	1	6.5	12.5
2	9.5	1	4	9
3	7	1	3	6.5
4	5	1	-	4.5
5	3.5	1	-	3
6	10	1	5	9.5
7	7	1	3.5	6.5
8	5	1	-	4.5
9	2.5	1	-	2
10	1.5	0.5	-	1
11	5	1	-	4.5
12	3	1	-	2.5
14	3.5	1	-	3

Dash (-) indicates mid depth samples not collected.

current measurements were made at Stations 2, 6, 7, 8, and 9 with drogues. The drogues consisted of two pieces of notched sheet aluminum, approximately 0.5 m on a side, which were inserted into one another to form a cross. Each drogue was attached with fishing line to a small surface float; the length of line used determined the depth at which the current was measured. The velocity of the current was determined from the anchored sampling vessel by measuring the rate of movement of the surface float with a stop watch and hand-held split image range finder. The direction of movement was measured using a compass. At Stations 2, 6, and 7, currents were measured at the surface, 3 m, and 5 m; however, due to the shallowness of Rowlett Arm, only surface current measurements were made at Stations 8 and 9. The wind velocity and direction at the west side of the Ray Hubbard dam were measured by DWU.

Samples were collected from each of the tributaries to Lake Ray Hubbard and the two outlets (the spillway ard the East Side Water Treatment Plant intake) at weekly intervals during the summer of 1976 and every two to four weeks during the fall, winter, and spring and every two weeks during the summer of 1977. The samples for the chemical analyses were collected with a Van Dorn sampler at a depth of 3 m at Station 13 (the intake) and at 6 m at Station 22 (the spillway). Since most of the other streams were too shallow to use a Van Dorn sampler, a bucket and rope were used for collection of a surface sample at each of the other tributary stations and of the Garland-Rowlett sewage treatment plant effluent.

Temperature and DO were measured with a YSI Model 57 DO meter at the surface at each tributary station, and at 1 m intervals over the water column at the East Side Water Treatment Plant intake.

All samples were stored on ice from the time of collection until they were delivered to the DWU and UTD laboratories. At the laboratories, the samples were stored in the dark at 4°C until the analyses were performed.

In addition to the regular sampling, at quarterly intervals, samples were collected from the lake and tributaries for determination of the concentrations of sulfate, copper, nickel, zinc, cadmium, chromium, lead, iron, manganese, mercury, calcium, sodium, and potassium.

Primary productivity studies using the light and dark bottle DO method were conducted in each of the four areas of Lake Ray Hubbard on July 30, August 19, September 23, 1976, and March 18, 1977. Light and dark bottles were set at 0.5 m, 1.5 m, and 2.5 m at Stations 3, 6, and 11. Since the depth of Station 9 was 2 m, bottles were set at 0.5 m and 1.5 m. Net photosynthesis was determined by measuring the change in the DO concentration in the light bottles after four hours of incubation. The amount of carbon fixed/m^3/hr was calculated for each depth at which bottles were set by the following equation:

$$\text{mg C fixed/m}^3\text{/hr} \quad \frac{\text{Net Photosynthesis as } \frac{\text{mg } O_2}{\ell} \, \frac{12 \text{ mg C}}{32 \text{ mg } O_2} \, \frac{1000 \; \ell}{\text{m}^3}}{\text{Incubation Time in Hours}}$$

This was integrated over the euphotic zone to obtain mg C fixed/m^2/hr. The euphotic zone was defined to be the depth to which one percent of the surface light penetrates. Light intensity was measured with a submarine photometer. The one percent depth is the point at which photosynthetic oxygen production equals respiration. This varied from station to station and over time at each station. When the euphotic zone was greater than 3 m, it was assumed that the productivity decreased linearly from 3 m to the bottom of the euphotic zone.

ANALYTICAL METHODS

Nutrient concentrations and other chemical parameters were measured by the DWU Research Laboratory, generally with procedures outlined in *Standard Methods*, Fourteenth Edition (APHA *et al*., 1976) and the US EPA *Manual of Methods for Chemical Analysis of Water and Wastewater* (US EPA, 1974). Listed below are the analytical procedures used by the DWU laboratory for the various chemical analyses. All spectrophotometric measurements were made with a Bausch & Lomb Spectronic 20 colorimeter.

Total phosphorus was determined by the ascorbic acid method (US EPA, 1974, p. 249). The samples were digested with persulfate. A 2.54 cm light path length was used.

Soluble orthophosphate was measured by the ascorbic acid method without digestion, after filtration through a 0.45 μ pore size membrane filter, using procedures presented by the US EPA (1974, p. 249).

Nitrate was determined by the Brucine method (APHA *et al*., 1976, p. 427) after filtration through fiberglass filters.

Nitrite analyses were performed on fiberglass-filtered samples using the diazotation method (US EPA, 1974, p. 215).

Ammonia determinations were made on unfiltered samples using the Orion specific ion electrode, model 95-10 until January 1977, using the standard addition procedure (US EPA, 1974, p. 165). After January 18, 1977, ammonia analyses were performed by the Nesslerization method (APHA *et al*., 1976, p. 407). The sample was distilled into boric acid.

Organic nitrogen was measured as the difference between total Kjeldahl nitrogen and ammonia initially present in the sample (APHA *et al*., 1976, p. 437). For the period up to January 18, 1977, the ammonia produced by the digestion was measured by the specific ion electrode method.

Alkalinity was determined on fiberglass-filtered samples by titration with sulfuric acid in accord with the procedure presented in *Standard Methods* (APHA *et al*., 1971, p. 52).

Hardness was determined on a fiberglass-filtered sample with EDTA titration (APHA et al., 1976, p. 202).

Chloride was determined using the mercuric nitrate titrimetric procedure presented in Standard Methods (APHA et al., 1976, p. 304). On September 28, 1976, the sample volume was increased from 20 to 100 ml.

Sulfate was determined by addition of excess barium chloride in which the unprecipitated barium was determined by atomic absorption spectrophotometry, following the procedures in the "Atomic Absorption Operations Manual" Perkin Elmer Corporation, March 1971.

Suspended solids were determined according to the procedure presented in Standard Methods (APHA et al., 1971, p. 537) using Reeve Angel fiberglass filters.

pH was measured on an Orion 801A Analyzer in accord with the procedures presented in Standard Methods (APHA et al., 1976, p. 460). All pH measurements were made at room temperature.

Turbidity was measured on a HACH model 2100 turbidimeter using formazin standards.

Heavy metals (Cu, Ni, Zn, Cd, Cr, Pb, Fe, Mn, Hg) and Ca, Na, and K were determined using the APDC-MIBK complexation and extraction method (US EPA, 1974, p. 78), "Water Analysis by Atomic Absorption," Varian Techtron, 1972, p. 13, "Methods for Collection and Analysis of Water Samples for Dissolved Minerals and Gases," USGS Tech. Water Resources Inv., Book 5 (1970).

Listed below are the analytical procedures used by the DWU East Side Water Treatment Plant Laboratory for the biological analyses.

Algal counts and identification were made on samples concentrated by filtration through 0.45 μ size membrane filters according to procedures presented in Standard Methods (APHA et al., 1976, p. 124, p. 1131). The filter was washed with 3 mls algae-free water until sediment from the filter was visually in suspension. One ml was then pipetted into a Sedgewick-Rafter Counting Cell. Microscopic examination was done with a phase contrast microscope at 200 X magnification. Algae were identified to the generic level using "Key for Identification of Freshwater Algae" (APHA *et al.*, 1976) and *Freshwater Biology* (Ward & Whipple, 1959).

The number of actinomycetes per ml was determined according to the procedure presented in *Standard Methods*, (APHA *et al.*, 1976, p. 987).

The threshold odor number was determined according to the procedure presented in *Standard Methods* (APHA *et al.*, 1976, p. 75) and the odor characteristic was determined according to the procedure presented in *Standard Methods* (APHA, *et al.*, 1965) with several modifications. Normally, the T.O.N. and odor characteristic were determined at 60°C by one observer.

Listed below are the analytical procedures used at UTD for the various chemical and biological parameters measured in the UTD laboratory:

Total phosphorus was measured after persulfate digestion by the ascorbic acid method outlined in *Standard Methods*, 13th edition (APHA *et al.*, 1971, p. 526). The following modifications of the persulfate digestion were made: a 20 ml aliquot was used and 0.2 ml of sulfuric acid solution and 3 ml of potassium persulfate solution were added. The following modifications of the ascorbic acid procedure were made: the alcohol treatment was eliminated, samples were allowed to stand for 30 minutes before the transmittance was read on a Beckman DU spectrophotometer using a 2 cm light path.

Soluble orthophosphate was measured by the ascorbic acid method (APHA *et al.*, 1971, p. 526) with the modifications described for total P.

Ammonia was measured with an Orion Model 801A Analyzer and an ammonia ion electrode (US EPA, 1974, p. 165).

Nitrate was measured by the procedure presented in the US EPA manual (US EPA, 1974, p. 197). The procedure was also automated as outlined in the US EPA manual (US EPA, 1974, p. 207) and run on the Technicon Auto-Analyzer I.

Nitrite was measured by the procedure given in the US EPA manual (US EPA, 1974, p. 215). The procedure was also automated and run on the Technicon Auto-Analyzer I.

pH was measured with a pH electrode with an Orion Model 801A Analyzer (APHA _et_ _al_., 1976, p. 460).

Specific conductance was determined with a model 33 YSI SCT meter which automatically corrects conductance to a temperature of 25°C (APHA _et_ _al_., 1976, p. 71).

Chlorophyll _a_ and pheophytin _a_ were measured by the spectrophotometric methods presented in _Standard_ _Methods_ (APHA _et_ _al_., 1976, p. 1029). This involved the use of a Beckman model DU spectrophotometer with a 5 cm light path.

QUALITY CONTROL PROGRAMS

No laboratory quality control program was used by Dallas Water Utilities in their analysis of samples. In general, their analytical results represent single analyses of the samples. No spiked samples were analyzed. It was assumed without verification that the analytical methods used were applicable to these water samples. No attempt was made to determine if there were interferences in the samples which would make the analytical procedure not applicable.

The analytical quality control program used in the UTD laboratory involved running triplicate analyses for total P, ammonia, nitrate, and nitrite determinations. Duplicate analyses were run for soluble ortho P, chlorophyll _a_,and pheophytin _a_ determinations. Specific conductance and pH were determined once for each sample.

ESTIMATED STREAM DISCHARGE AND NUTRIENT LOADS

Dallas Water Utilities provided daily estimates of the flow of Muddy Creek, Rush Creek, Cottonwood Creek, Camp Creek, Squabble Creek, and Yankee Creek based on a Hydrocomp hydrologic model of the Lake Ray Hubbard watershed. The flows for Rowlett Creek and the East Fork of the Trinity River (below Lake Lavon) were obtained from the USGS which maintains gaging stations at these sampling stations. The Garland wastewater treatment plant (Station 24) furnished daily flows from the plant records. The total input from the tributaries to the lake and the output from the lake were calculated from the daily estimates of flow and the average of the nutrient concentrations between sampling dates. Further information on the approach used to calculate nutrient loads is provided in the Results Section of this report.

Nutrient input from the atmosphere was estimated from data obtained from dust and precipitation collection buckets placed in the watershed; one on the east side of the Lake Ray Hubbard dam, one on the west side of the dam, and one in Elgin Robinson Park (Figure 7). Two other precipitation and dustfall buckets were located in Parker, Texas, about 16 km north of Lake Ray Hubbard,and one other bucket was located in Denison, Texas about 120 km north of the lake on the property of M. Gary. It was felt that the additional buckets would furnish a regional estimate of the nutrient loading from the atmosphere. Atmospheric loads were calculated from the amounts of nutrients that fell into the buckets which had a diameter of 26.7 cm and an area of 0.056 m^2. The buckets were placed on a pedestal approximately 1.5 m above the ground. A circle of plastic clothesline with a diameter 10 cm greater than the diameter of the top of the bucket was suspended approximately 6 cm above the top of the bucket. This provided a bird roost which minimized contamination of the samples by bird fecal material.

If the nutrients were not in solution from the previous precipitation, a known volume of water was added to the bucket and the concentrations of the nutrients were measured by the procedures previously described. Since there was occasionally contamination from bird fecal material, only buckets thought to be uncontaminated, based on visual inspection, were used for nutrient measurements. Nutrient concentrations that appeared to be anomalous were statistically evaluated by the Dixon Criterion (Natrella, 1963), and the values failing the test were rejected. Atmospheric loading was expressed as a yearly loading coefficient in $kg/m^2/yr$. The loads were calculated by multiplying the nutrient concentration by the volume of precipitation or washwater, dividing by the number of days of collection, and multiplying or dividing by the appropriate conversions. This approach provided only a crude estimate of the atmospheric-derived nutrient loading to Lake Ray Hubbard.

SECTION 6

RESULTS

The results of this study may be conveniently divided into several different areas: nutrient loads, general physical and chemical characteristics, nutrients (nitrogen and phosphorus compounds), biological response parameters, and water quality characteristics. The results obtained in the studies of each of these areas are presented in this section of the report. Primary emphasis is given to the characteristics of the Main Body of the lake and the factors influencing these characteristics. The characteristics of the East Fork Arm, Muddy Creek Arm, and Rowlett Arm are discussed and compared to the Main Body. A discussion of the general trends in the data is presented in this section of the report. In addition, where possible, a discussion is presented of the factors which seem to be controlling the trends in the data. The discussion of the overall significance of these data as they relate to water quality management in Lake Ray Hubbard and other water bodies is presented in the discussion section of this report.

For each major area of the lake (Main Body, Rowlett Arm, East Fork Arm, and Muddy Creek Arm), the data from all sampling stations were averaged. During the periods of thermal stratification, the surface and mid-depth data from the Main Body stations were averaged and the mean concentration was plotted. In the shallower arms of the lake, mid-depth samples were not collected so the surface concentration was plotted. During the months when the water column was isothermal, the bottom depth data were also included in the averaging. The data from Station 14 were not included in the averaging of Rowlett Arm data since these data were not representative of the rest of the arm because this station is near the discharge of the Garland sewage treatment plant.

In order to obtain the average pH values, the pH was converted to the hydrogen ion concentration and the arithmetic mean hydrogen ion concentration was determined. This average value was then converted to pH. The average Secchi depth was found by converting all Secchi depth values to log Secchi depth. In most of the figures of chemical data presented in this section, dashed lines were used to indicate that there was uncertainty about the shapes of the curves between the data points.

CHARACTERISTICS OF THE MAIN BODY

Dissolved Oxygen and Temperature

Lake Ray Hubbard is a warm water lake which does not retain a permanent ice cover during the winter months. The summer surface water temperatures are generally about 10°C higher than the temperatures of northern lakes. They ranged from 23°C in May and early June to 32°C in July and August. Figure 8 presents the surface water temperatures between June 1976 and August 1977 for each of the four major areas of the lake. The surface temperature started decreasing in September and reached a minimum of 4°C in January when there was a thin (5mm) sheet of ice over much of the lake for several days. The temperatures increased during the spring and summer months and reached a maximum near 32°C in August of both years.

The winter of 1976-1977 was unusually cold. Previous studies by the Texas Water Quality Board showed water temperatures of 8°C in January (Personal Communication, J. Pettitt, 1977). The maximum surface temperatures found during the two summers (1976 and 1977) were about the same as temperatures found by the Texas Water Quality Board during the previous years. Temperatures above 30°C may be of ecological significance since this is above the temperature for optimum growth of some planktonic organisms (US EPA, 1976). The nutrient regimes of southern lakes could be markedly different from those of northern lakes due to warmer

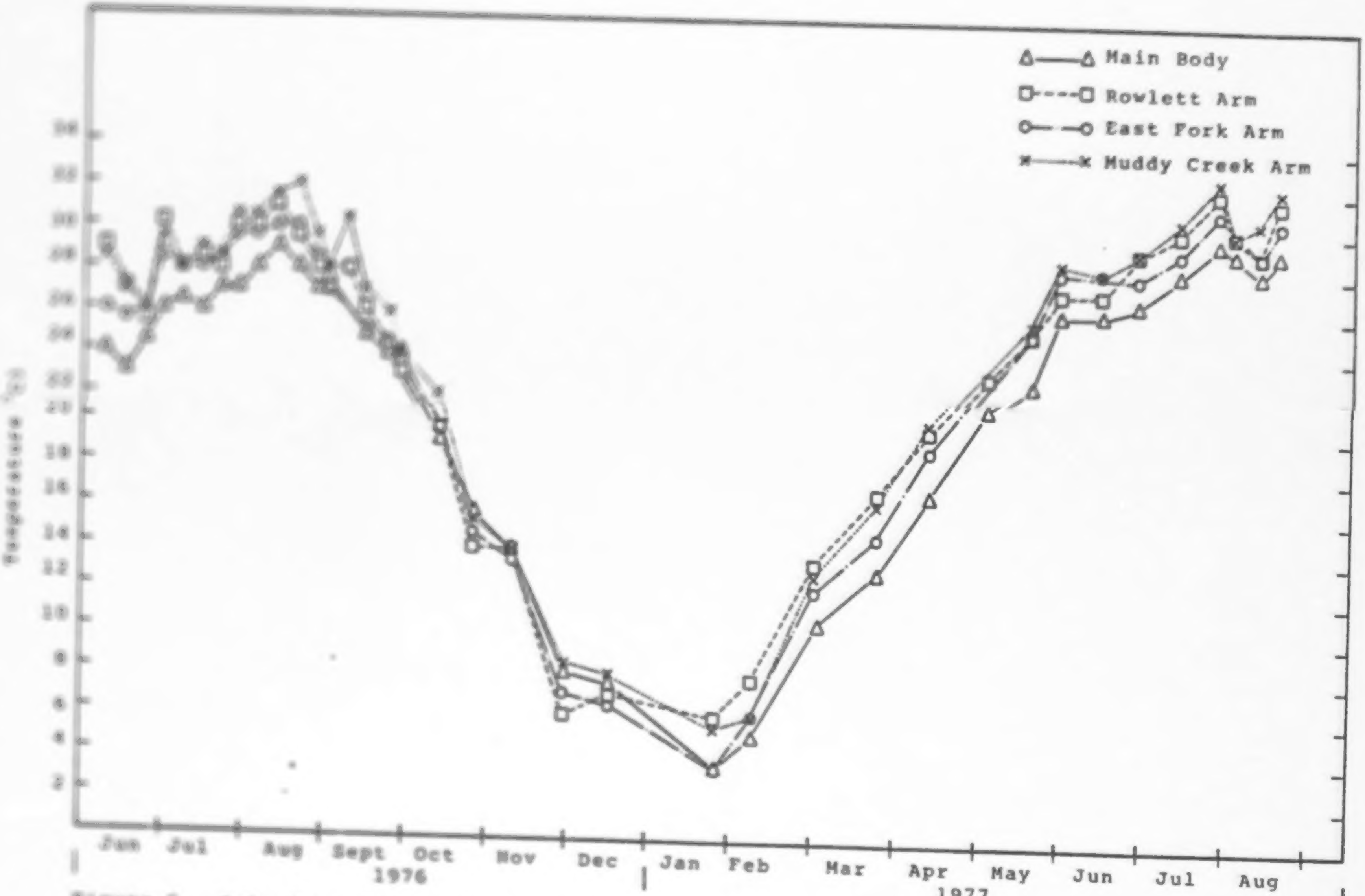


Figure 5. Lake Ray Hubbard surface water temperatures

water temperatures during the winter months. This would promote more rapid regeneration of nutrients during the winter in southern lakes than in northern lakes.

In the Main Body, a thermocline started to develop in May in the deeper areas and became permanently established near 10 m during the summer months of 1976 and 1977. Figure 9 is a depth-time diagram of isotherms at Station 1. The figure shows that this part of the Main Body was isothermal from mid-September to May, indicating that the water column was well mixed during this period of time. A depth-time diagram of isopleths of dissolved oxygen (DO) at Station 1 is shown in Figure 10 . During the summer months, the DO was completely depleted below 10 m due to the sediment oxygen demand and the decomposition of organic matter (predominantly algae) sinking from the surface waters through the thermocline which served as an effective barrier to mixing of hypolimnetic water with oxygen-rich epilimnetic waters. The thermocline disappeared in mid-September, the water column became completely mixed, and the DO concentration became constant from surface to bottom. This pattern persisted throughout the winter and spring months until the thermocline became established again in May. Figure 11 presents several DO and temperature profiles at Station 1. The DO profiles show the same trends as the temperature profiles, being constant from surface to bottom during the winter and changing rapidly with depth during the summer. During the winter months, the DO concentration was at, or slightly greater than, the saturation concentration. However, throughout most of the rest of the year, the surface DO concentrations were about 1 mg/l below the saturation concentration. This is unusual since highly productive lakes usually have supersaturated surface waters due to the photosynthetic production of oxygen.

The other Main Body stations were shallower than Station 1 and had transient thermoclines which were present during some weeks of the summer and absent during others, depending on the degree of wind mixing. The thermocline was never established for a period of sufficient duration to become a permanent, effective barrier to mixing of surface and bottom waters.

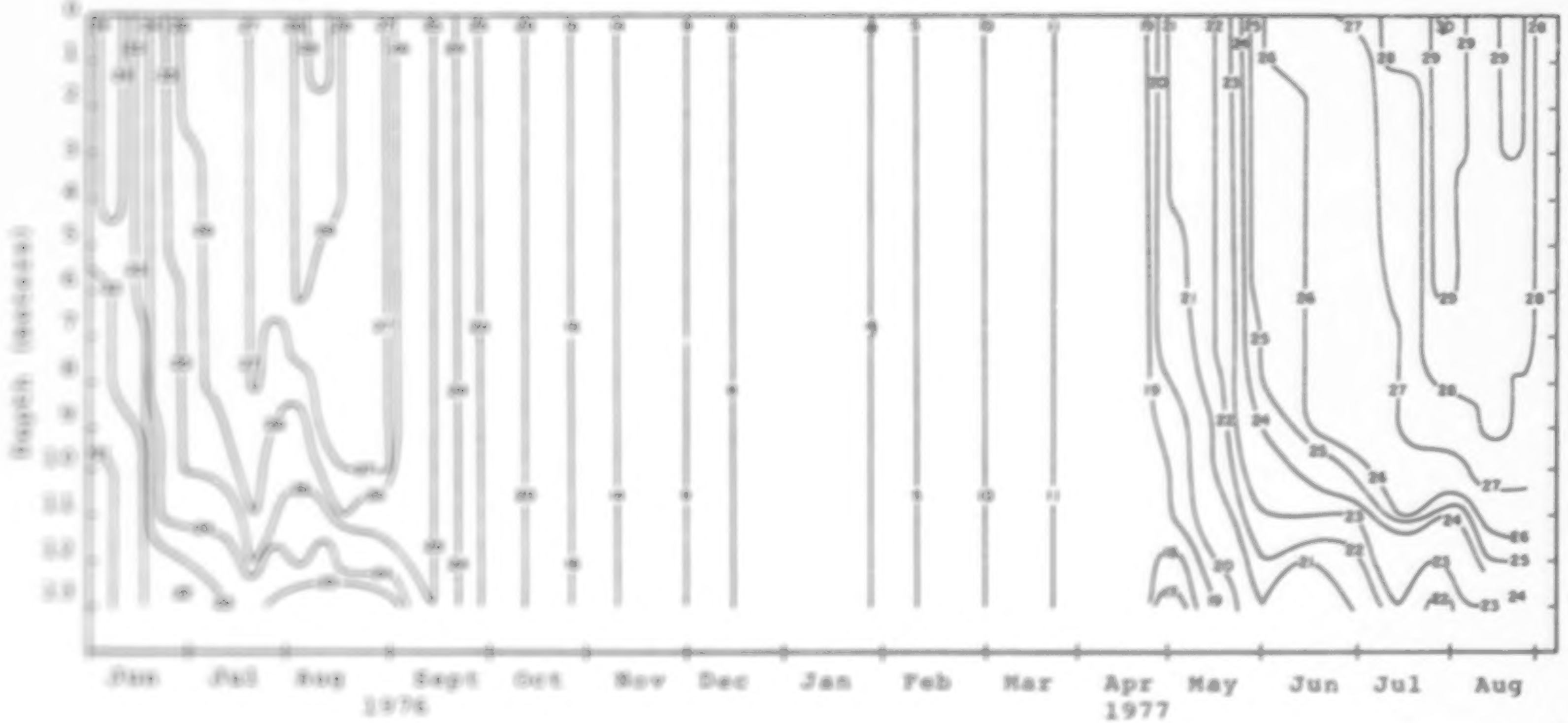


Figure 9. Isotherms - Station 1

(Values reported in °C)

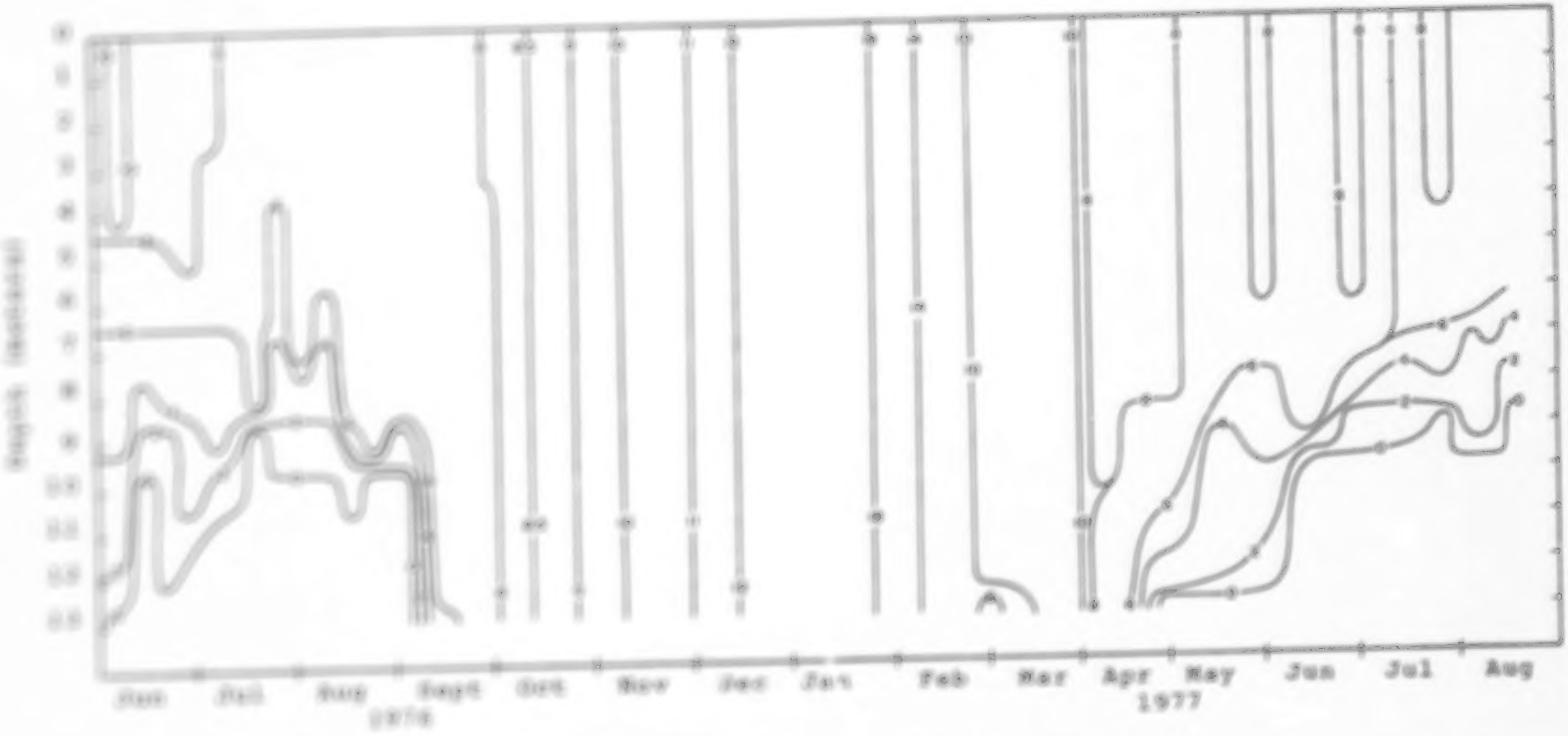


Figure 13. Isopleths of dissolved oxygen - Station 1
(Values reported in mg/l)

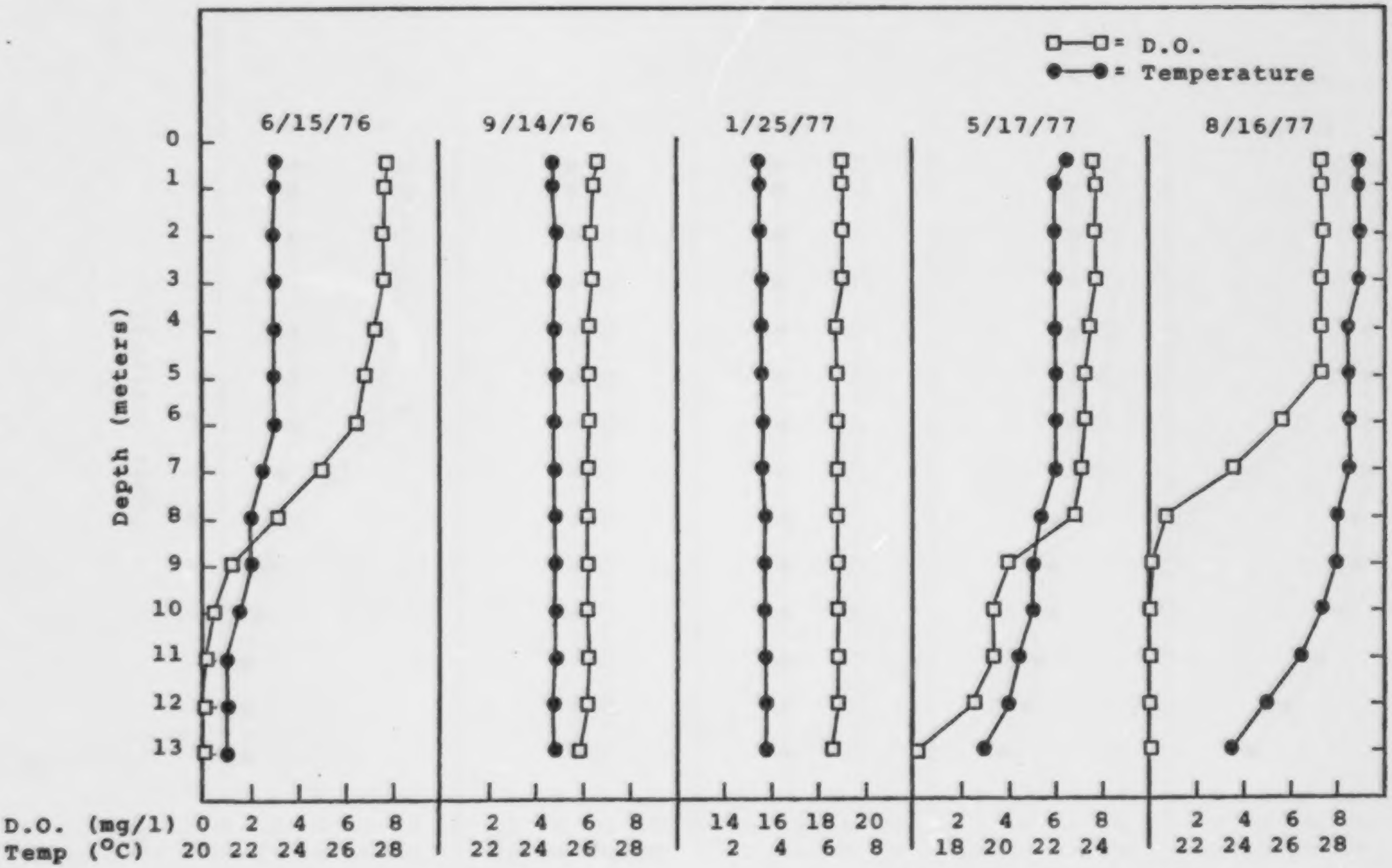


Figure 11. Dissolved oxygen and temperature profiles - Station 1

Complete depletion of DO was found at Station 1 throughout the summers of 1976 and 1977. Intermittent periods of complete DO depletion occurred in the bottom water at Stations 2 and 6 during the summer months of both study years. However, complete depletion of DO was not found at Station 7.

Phosphorus

The average total phosphorus and the average soluble ortho P concentrations in the Main Body are shown in Figures 12 and 13, respectively. The total P concentration was generally near 0.03 mg-P/l with peaks in June 1976 and March 1977. There were small peaks in August (0.09 mg-P/l) and October (0.06 mg-P/l) 1976. The peak of 0.28 mg-P/l in June was likely associated with P contained in algal cells since the algal population was large at this time, and the soluble ortho P concentration was only 0.003 mg-P/l. The small peak in August was also likely due to the algal incorporated phosphorus. The algal population was small during October so the soluble ortho P was not rapidly removed from the water column. The peak of 0.12 mg-P/l in late March was likely due to the large input of P from the tributaries during February and March.

The soluble ortho P concentration was variable over time, decreasing to less than the detection limit (a few µg-P/l) at times and reaching peaks of 0.014 mg-P/l in October 1976 and 0.020 mg-P/l in May 1977. The fluctuations which occurred in soluble ortho P during the summer of 1976 were most probably due to analytical problems associated with the analysis of soluble ortho P at concentrations of 10 µg/l P or less. The highest soluble ortho P concentrations were found during the periods of low algal numbers, and the low concentrations were found during the algal blooms when essentially all of the soluble ortho P had been incorporated into algal cells and was limiting algal growth. An inverse relationship between the soluble ortho P concentration and the algal numbers would be expected in a lake where P is the algal growth limiting nutrient.

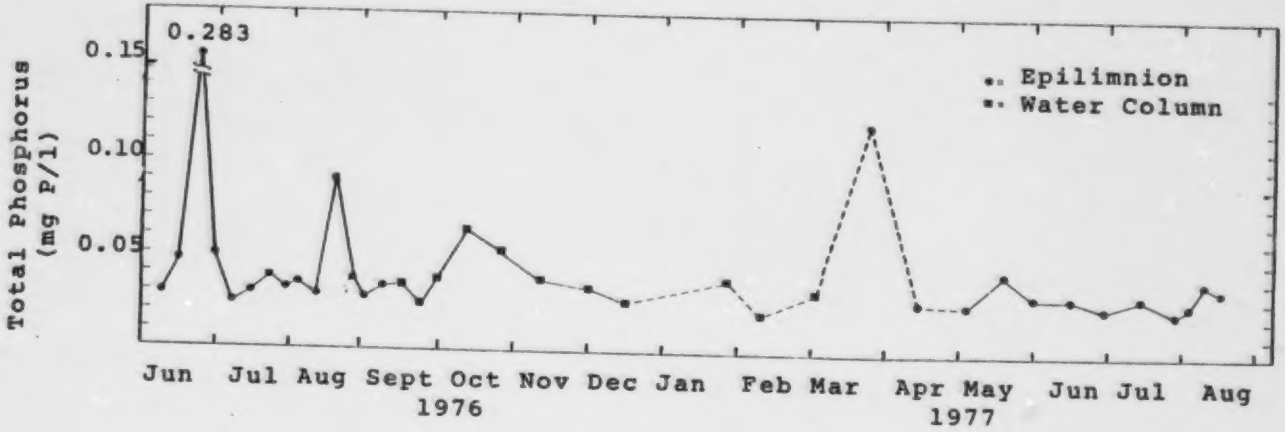


Figure 12. Average total phosphorus concentration - Main Body

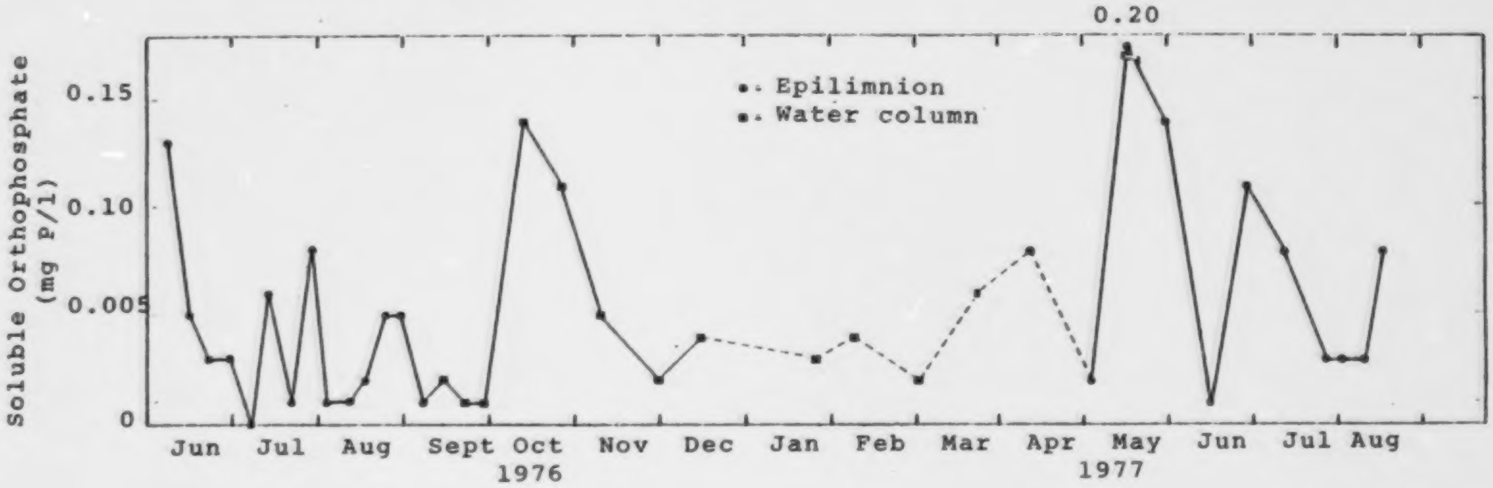


Figure 13. Average soluble orthophosphate concentration - Main Body

The very low levels of soluble ortho P (i.e., a few μg/l) found during certain times of the year, coupled with high algal concentrations at the same time, are indicative of the possibility of P limitation of algal growth. The overall levels of total phosphorus and soluble ortho P are indicative of fertile waters which would normally be classified as eutrophic.

Nitrogen

The average total N and the average inorganic N concentrations in the Main Body are shown in Figures 14 and 15. The total N concentration was generally between 0.5 and 1.0 mg-N/l with peaks during June and November 1976 and March and April 1977. The June 1976 peak of 3.7 mg-N/l was most likely due to the large influx of nutrients from the tributaries during the spring and the nitrogen incorporated into algal cells. The peak of 2.1 mg-N/l during November 1976 occurred when the lake was isothermal and was likely due to release of N from the sediments since there was a corresponding increase in the inorganic N concentration at this time. The high concentrations of total N found during March and April 1977 (1.1 to 1.7 mg-N/l) reflect the large influx of nutrients from the tributaries during February, March, and April.

The inorganic N concentration generally ranged from 0.04 to 0.12 mg-N/l with two major peaks during the study period in June 1976 and April to mid-June 1977. Since soluble ortho P was generally in lower concentrations relative to the stoichiometric requirements of algae, inorganic N was most likely not limiting algal growth through most of the study period in this part of the lake. The peak of 0.54 mg-N/l during June 1976 and the high concentrations of 0.34 to 0.60 mg-N/l found during the spring of 1977 were likely due to inputs of nutrients from the tributaries.

Nitrogen to Phosphorus Ratio

Most algae take up phosphorus in the form of soluble orthophosphate and nitrogen in the form of nitrate or ammonium. The inorganic nitrogen ($NO_3^-+NO_2^-+NH_4^+$) concentration measures the

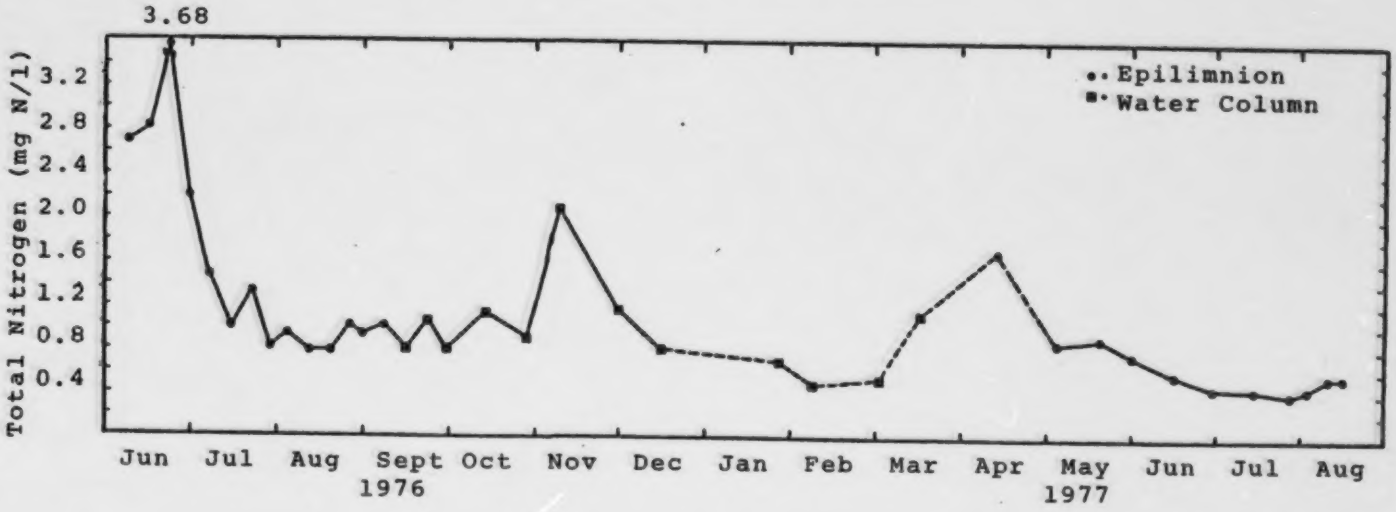


Figure 14. Average total nitrogen concentration - Main Body

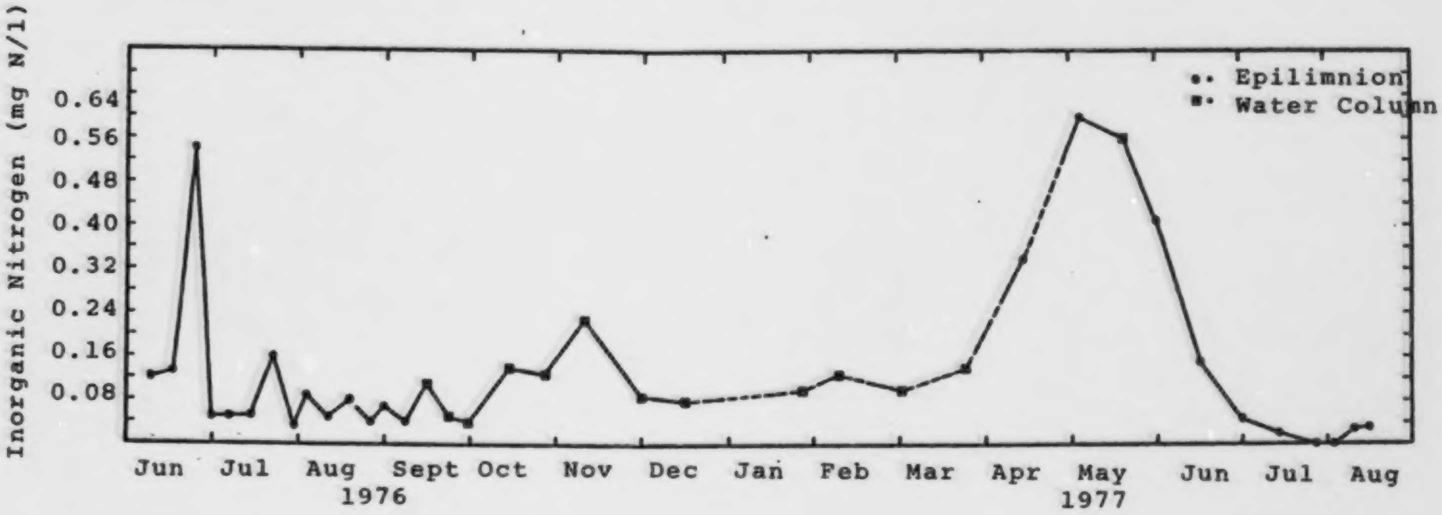


Figure 15. Average inorganic nitrogen concentration - Main Body

nitrogen readily available for use by many types of algae which use approximately 7 grams of nitrogen for every gram of phosphorus. The ratio of inorganic N to soluble ortho P is indicative of the algal growth limiting nutrient (Lee, 1973a). A N:P ratio greater than about 7 indicates that there is excess phosphorus, while a ratio greater than 7 indicates excess nitrogen relative to the normal stoichiometric requirements of algae.

Caution must be exercised in using these numbers to determine whether N or P is limiting in a particular water sample. In order for one of the elements to be limiting, the concentrations in the water must be relatively low compared to organism needs for further growth. For phosphorus, this is generally accepted to be a few µg-P/l. For inorganic nitrogen, the critical level is generally accepted to be less than 0.1 mg-N/l. If both of these elements are present in concentrations somewhat larger than these amounts, then neither of these elements are limiting algal growth, independent of the N:P ratio. N:P ratio data should not be extrapolated when there is a surplus of N and P, since it is highly likely that the N:P ratio will change significantly during the development of an algal bloom as a result of different rates of supply of the two elements to the water body.

As shown in Figure 16 , the N:P ratios in the Main Body were typically greater than 7, indicating that phosphorus may have been limiting algal growth in this part of the lake. There were times, however, during the summer when the N:P ratio approached the normal stoichiometric requirements of algae. Under those conditions, both N and P may have been limiting the rate of growth and the total biomass of algae present in the Main Body of the lake.

pH

The average pH was calculated for the Main Body stations at the surface, mid, and bottom depths for the periods of thermal stratification (June through September 1976 and June through August 1977) and at the surface and bottom depths for the isothermal period (October 1976 through May 1977). The data

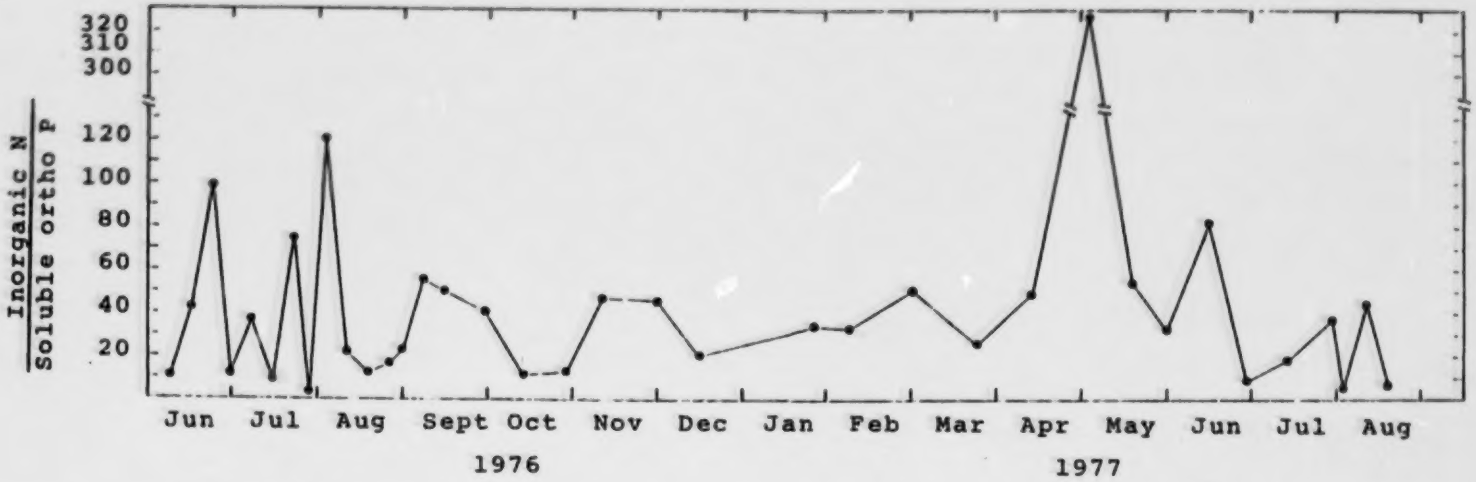


Figure 16. Inorganic nitrogen to soluble orthophosphate ratios - Main Body

presented in Table 11 show that the pH is essentially constant throughout the Main Body during each period.

Figure 17 shows the pH in the surface and bottom waters at Station 1. During the periods of thermal stratification, the surface pH was normally higher than the bottom pH, most likely due to the photosynthetic uptake of CO_2 by algae in the surface waters and respiration of bacteria in the bottom waters. The pH ranged from 7.2 to 8.3 during the summer of 1976, with the surface pH generally being near 8.0. However, during the summer of 1977, the pH ranged from 7.5 to 8.5 with the surface pH near 8.5. The higher pH values found during the summer of 1977 were likely due to the lower solubility of CO_2 in the surface waters as a result of the warmer water temperatures during this summer. As shown in Figure 17 , the surface waters during the summer months of 1977 were, in general, about 2°C higher than during the summer of 1976. Since the solubility of CO_2 decreases with increasing temperature, it would be expected that the pH during the summer of 1977 would be higher than during the summer of 1976. During the winter months, when the water column was isothermal and completely mixed, the surface and bottom waters typically had a pH of 8.0.

In order to evaluate the possible significance of the difference in temperatures of the two summers on the CO_2 content of the water, a series of computations was made to assess the free CO_2 content of the water during the two summers. Using approximate average values for pH, specific conductance, alkalinity, and temperature, it was found that the CO_2 content of the surface waters during the summer of 1976 was approximately 1.4 mg/l. The corresponding value for the summer of 1977 was 0.9 mg/l free CO_2. Calculations of the free CO_2 content of the bottom waters showed that in the summer of 1976, it was approximately 6.5 mg/l. The bottom water free CO_2 content during the summer of 1977 was approximately 1.0 mg/l. Since the pH of a water that is buffered with the CO_2-HCO_3^- system is highly dependent on the CO_2 content, it, therefore, appears plausible that the higher water temperatures

TABLE 11. AVERAGE pH - MAIN BODY

Depth	Station 1	2	6	7
6/76 to 9/76				
S	7.9	8.0	7.8	8.0
M	7.8	8.0	7.9	8.0
B	7.5	7.9	7.9	7.9
10/76 to 5/77				
S	8.0	8.1	8.0	8.2
B	7.9	8.1	8.1	8.1
6/77 to 8/77				
S	8.4	8.5	8.4	8.3
M	8.3	8.3	8.4	8.4
B	8.0	8.2	8.0	8.3

S = Surface (1 m)
M = Mid
B = Bottom (0.5 m from sediments)

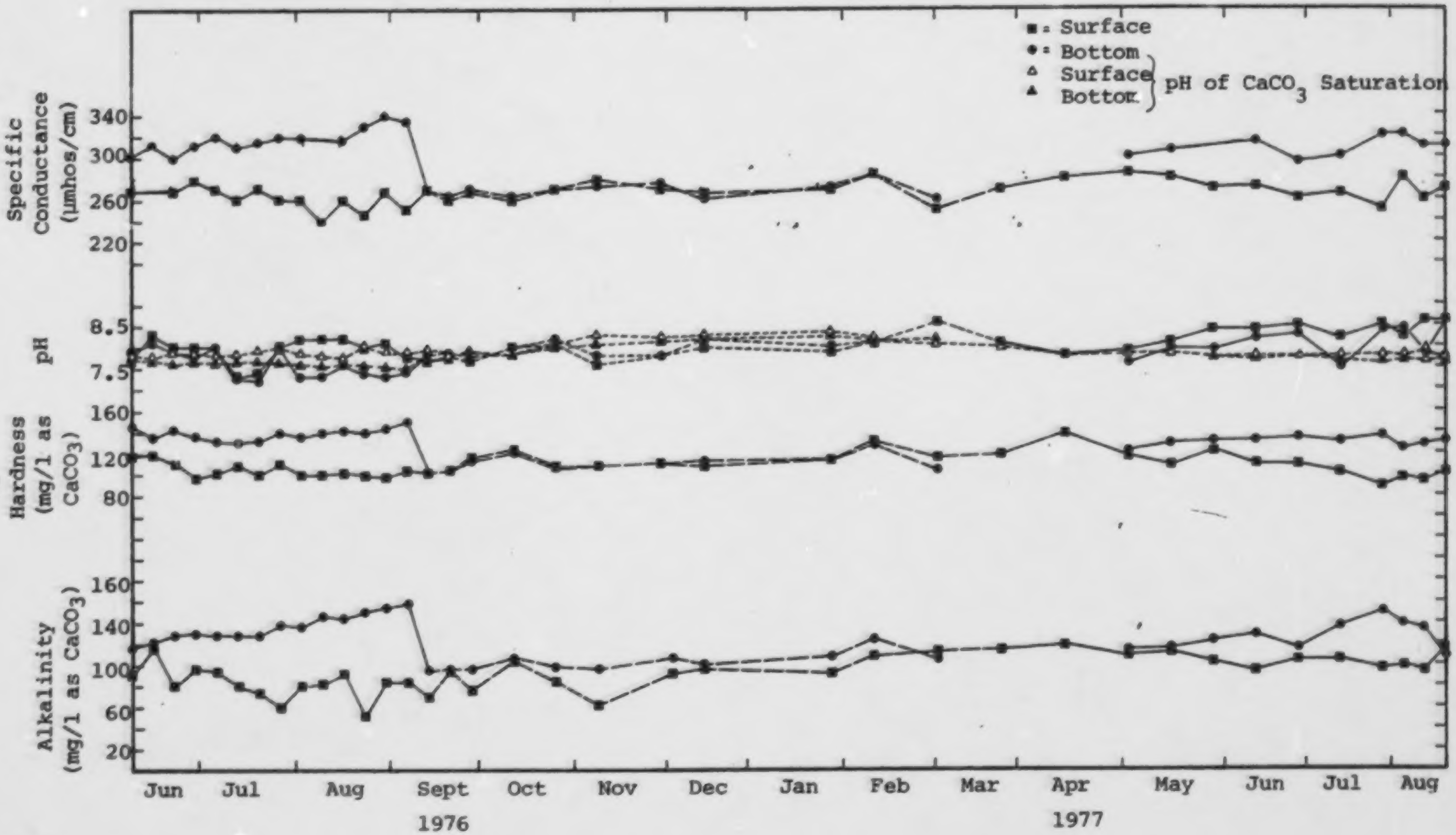


Figure 17. Chemical parameters - Station 1

during the summer of 1977, which reduced the CO_2 solubility, are the cause of the difference in the pH of the waters between the two summers.

Specific Conductance

The specific conductance of the surface and bottom waters at Station 1 is shown in Figure 17. All values presented in this figure are corrected to 25°C by the instrumentation used for measuring specific conductance. During the periods of thermal stratification, the specific conductance of the bottom waters was higher than that of the surface waters. During the summer of 1976, the specific conductance in the bottom waters ranged from 300 to 340 μmhos/cm, whereas at the surface, the range was 240 to 280 μmhos/cm. During the summer of 1977, the range was 295 to 320 μmhos/cm in the bottom waters and 250 to 285 μmhos/cm in the surface waters. The higher values at the bottom were likely due to the greater amounts of dissolved solids resulting from the dissolution of $CaCO_3$ from the sediments. During the winter months, the specific conductance of the surface and bottom waters was equal and ranged from 250 to 285 μmhos/cm. While dissolved solids were not measured in this investigation, they may be estimated from the specific conductance values using the relationship:

$$\underset{(\mathrm{mg/l})}{\text{Dissolved Solids}} = 0.7\ \underset{(\mu\mathrm{mhos/cm})}{\text{(Specific Conductance)}}$$

This relationship has been found in many natural waters (APHA *et al.*, 1976). From this relationship, it was estimated that the dissolved solids in Lake Ray Hubbard ranged from 190 to 225 mg/1.

Hardness

The total hardness values in the surface and bottom waters at Station 1 are shown in Figure 17. During the summer of 1976, when the water column was thermally stratified, the hardness of the hypolimnion ranged from 135 to 150 mg/l as $CaCO_3$ and was higher than the hardness of the surface waters which ranged from 95 to 120 mg/l as $CaCO_3$. During the summer of 1977, the range of hardness in the bottom waters was 120 to 135 mg/l as $CaCO_3$, and the range in the surface waters was 90 to 120 mg/l as $CaCO_3$. The higher hardness in the hypolimnion indicates the possible dissolution of $CaCO_3$ from the sediments and from any $CaCO_3$ precipitation that occurred in the epilimnion. During the winter months, when the water column was isothermal, the hardness of the surface and bottom waters was near 110 mg/l as $CaCO_3$. Additional discussion of the precipitation of $CaCO_3$ is presented in the discussion section of this report.

Alkalinity

The alkalinity of the surface waters at Station 1 was lower than that of the bottom waters during the periods of thermal stratification, most likely resulting from the dissolution of $CaCO_3$ from the sediments and from the $CaCO_3$ raining down from the overlying waters. As shown in Figure 17, the alkalinity of the surface waters ranged from 50 to 115 mg/l as $CaCO_3$, and the bottom waters ranged from 115 to 140 mg/l as $CaCO_3$ during the summer of 1976. The range during the summer of 1977 was 95 to 115 mg/l as $CaCO_3$ in the surface waters and 115 to 150 mg/l as $CaCO_3$ in the hypolimnion. During the period when the water column was isothermal, the alkalinity ranged from 60 to 125 mg/l as $CaCO_3$ with the alkalinity of the bottom waters slightly higher than that of the surface waters.

This alkalinity pattern is compatible with the hardness and pH data in that there appears to be $CaCO_3$ precipitation from the surface waters during months and dissolution in the bottom waters. This pattern is to be expected in hard water eutrophic lakes and has been previously found by Lee and Delfino (1969) in Lake Mendota near Madison, Wisconsin.

Chloride

Chloride is a non-reactive or conservative chemical which should be indicative of mixing processes in the lake. Table 12

TABLE 12 AVERAGE CHLORIDE - MAIN BODY

Depth	Station 1	2	6	7
6/76 to 9/76				
S	21.2	16.5	17.5	16.0
B	16.0	15.8	16.4	17.0
10/76 to 5/77				
S	11.6	12.0	12.0	12.1
B	12.0	11.8	12.9	12.5
6/77 to 8/77				
S	15.3	14.4	14.3	15.3
B	14.7	14.9	14.2	15.3

S = Surface (1 m)
B = Bottom (0.5 m from sediments)

shows the average chloride concentrations at the Main Body stations for the periods of thermal stratification and the isothermal period. During the summer of 1976, the chloride concentration was relatively constant throughout the Main Body. With the exception of the surface concentration at Station 1, (21.2 mg/l) the range was from 15.8 to 17.5 mg/l. The concentration decreased during the winter months and ranged from 11.6 to 12.5 mg/l. During the summer of 1977, the concentration ranged from 14.2 to 15.3 mg/l which was lower than the concentrations found during the previous summer.

Although the chloride concentrations indicate there was dilution of the Main Body water by the high flows of low chloride concentration tributary water during the spring months of 1977, this dilution is not reflected in the alkalinity and hardness values which remained essentially the same during both summers. It would be expected that the alkalinity and hardness during the summer of 1977 would be lower due to the dilution. Since these parameters were not lower during the summer of 1977, $CaCO_3$ precipitation appears to be proceeding to the same residual alkalinity and hardness during both summers. The rate and degree of $CaCO_3$ precipitation in natural waters are influenced by a wide variety of factors, many of which are poorly understood at this time. It should be noted that the discussion of $CaCO_3$ behavior in various parts of Lake Ray Hubbard is predicated on an assumption that no colloidal $CaCO_3$ is present in the water. It is assumed that all hardness and alkalinity measured were due to chemical species in true solution since alkalinity and hardness samples were filtered through Reeve Angel fiberglass filters prior to analysis.

Secchi Depth

The Secchi depth is a measure of water clarity which is related to the amount of organic and inorganic turbidity in the water. Figure 18 shows the average Secchi depth and average surface turbidity in the Main Body during the study period. As

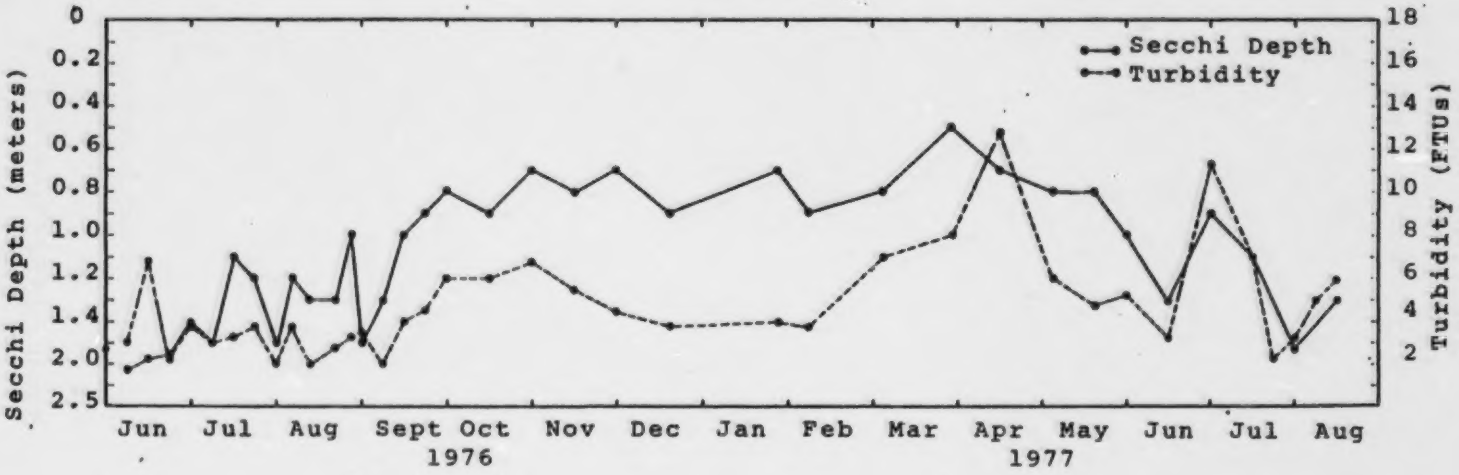


Figure 18. Average Secchi depth and turbidity - Main Body

shown in the figure, the Secchi depth and surface turbidity were inversely related. During the summer months when the lake was isothermal, the Secchi depth was greatest, ranging from 0.9 to 2.2 m. After the water column became completely mixed, the Secchi depth decreased and ranged from 0.5 to 0.9 m from September to May. The decreased Secchi depth during this time was likely due to the suspension of particulate matter from the bottom by wind and organism-induced mixing of the water. The turbidity increased from about 3 FTU's during the summer months to 4 to 7 FTU's during the winter months. The high peak in turbidity in mid-April 1977 was most likely related to the influx of turbid materials from the tributaries and the suspension of materials from the bottom into the overlying waters.

From the magnitude of the Secchi depth values, it is estimated that the euphotic zone, i.e., area where photosynthesis takes place, is approximately 2 to 6 m deep. These values are based on a relationship in which the Secchi depth multiplied by 4 is equal to the compensation depth.

The suspension of particulate matter from the bottom causes Lake Ray Hubbard to have a water clarity-Secchi depth pattern over the annual cycle different from that found in many other lakes and impoundments. Normally, lakes with fertility similar to Lake Ray Hubbard show relatively clear water during the late fall, winter,and early spring. In these water bodies, the clarity of the water is determined primarily by the numbers of algae present in the water. However, for Lake Ray Hubbard, water clarity is influenced by both algae and inorganic turbidity arising from the input from its tributaries and suspension from the sediments. For example, during the fall of 1976, the algal population in the Main Body was small, but the Secchi depths were only about 0.9 m. The Secchi depths in many other lakes with the same level of fertility would have been on the order of 3 to 4 m. It is evident that materials suspended from the sediments greatly decrease water clarity in the lake.

Light Penetration

The transparency of a water body is a function of the nature and density of suspended and dissolved matter in the water column. It is normally greatest during periods of low algal biomass and lowest during algal blooms. During the study, the transparency of the water fluctuated from week to week and was greatest during the summer months.

The Main Body had the clearest water in Lake Ray Hubbard during the study period. The average percent of the surface light intensity at a depth of 1 m ranged from 69 percent at Station 1 on 6/28/77 to a low of 3 percent at Station 7 on 7/12/77. The average value for the study period was 25 percent.

The compensation depth, where the photosynthetic rate equals the rate of respiration, determines the depth of the euphotic zone. This is usually defined to be the depth to which 1 percent of the incident light penetrates. At Station 1, the compensation depth was at 6 to 7 m during the summer months and at 4 to 5 m during the fall and winter months. This was the deepest part of the euphotic zone in the lake.

The Secchi depth is generally related to the depth to which 1 percent of the surface light penetrates, by an equation of the following form:

$$1\%\ \text{Light Depth (D)} = \text{Constant} \times \text{Secchi Depth}$$

In the early literature, this constant was frequently reported as 2.5. Recently, it has been determined to be closer to 4. In the Main Body of Lake Ray Hubbard, the factor relating the 1 percent depth and Secchi depth was found to be 3.6, based on one year's data collected from June 1976 through May 1977. This correlation is important because the Secchi depth can be measured easily, and the depth of the euphotic zone can be determined without light penetration measurements.

The extinction coefficient (Ke) is the slope of the line obtained by plotting the log of the percent light transmittance as a function of depth. It is a function of the inorganic and organic detrital turbidity that results from vertical mixing in the water column and the concentration of the phytoplankton in the water. The extinction coefficient values for Station 1 were fairly constant and ranged from 0.5 to 2 m^{-1} as shown in Figure 19. The figure shows that the higher extinction coefficients were found during the winter, most likely due to the greater amounts of suspended sediments in the water column during this season.

The Secchi depth is an easily measurable parameter of water quality, so a correlation between the Secchi depth and Ke allows the investigator to approximate Ke easily. Figure 20 is a plot of the extinction coefficients as a function of Secchi depth in Lake Ray Hubbard during the study period. The equation which describes this relationship in the Main Body of the lake is:

$$Ke = \frac{1.3}{(\text{Secchi})^{0.6}}$$

For many water bodies, the variability of the extinction coefficient with time is a function of the phytoplankton concentration. If there were no phytoplankton, there would still be extinction of light as it passed through the water column. The extinction coefficients in the four main areas of the lake have been correlated to surface uncorrected chlorophyll a concentrations. A correlation of the form Ke = Ke' + a x phytoplankton chlorophyll is obtained where Ke' is the extinction coefficient due to natural turbidity and other factors rather than chlorophyll a concentrations. These correlations explain the self-shading mechanism at high algal concentrations. The equation which describes the relationship between the extinction coefficient and the chlorophyll a concentrations in the Main Body is:

$$Ke\ (m^{-1}) = 1.1 + 0.017\ [\text{uncorrected chlorophyll } \underline{a}\ (\mu g/l)]$$

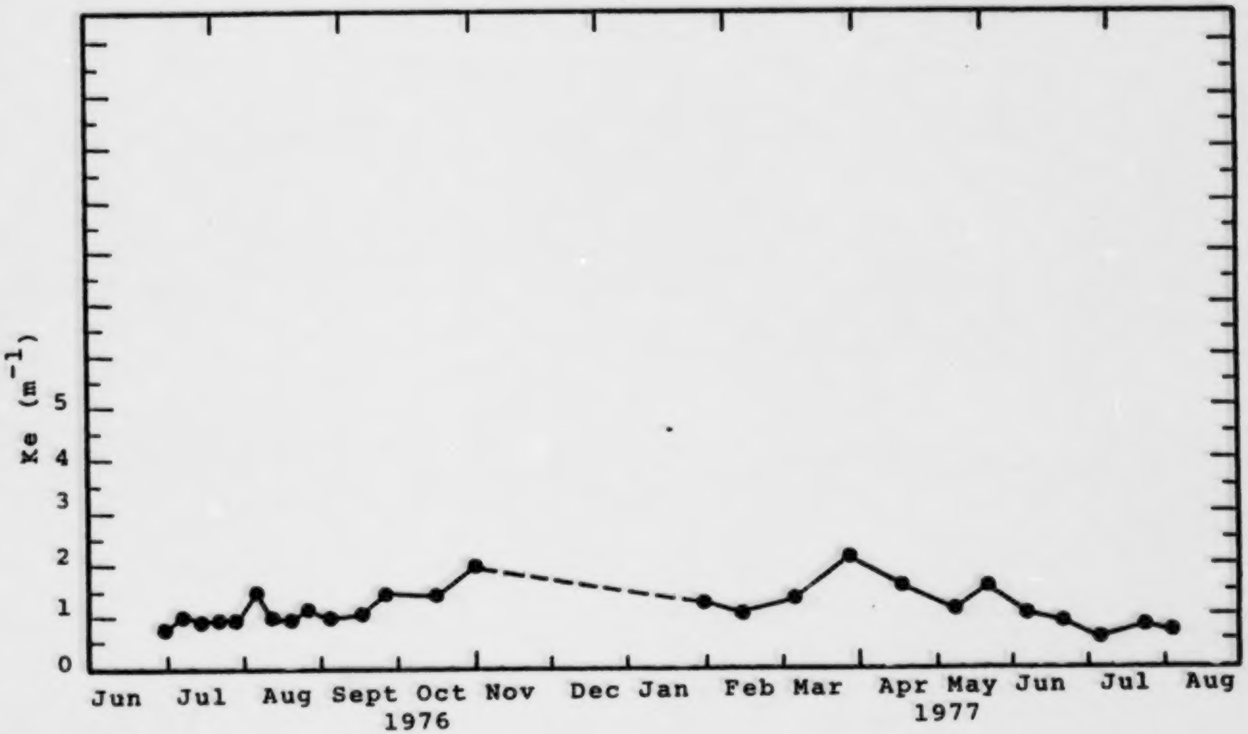


Figure 19 . Extinction coefficient - Station 1

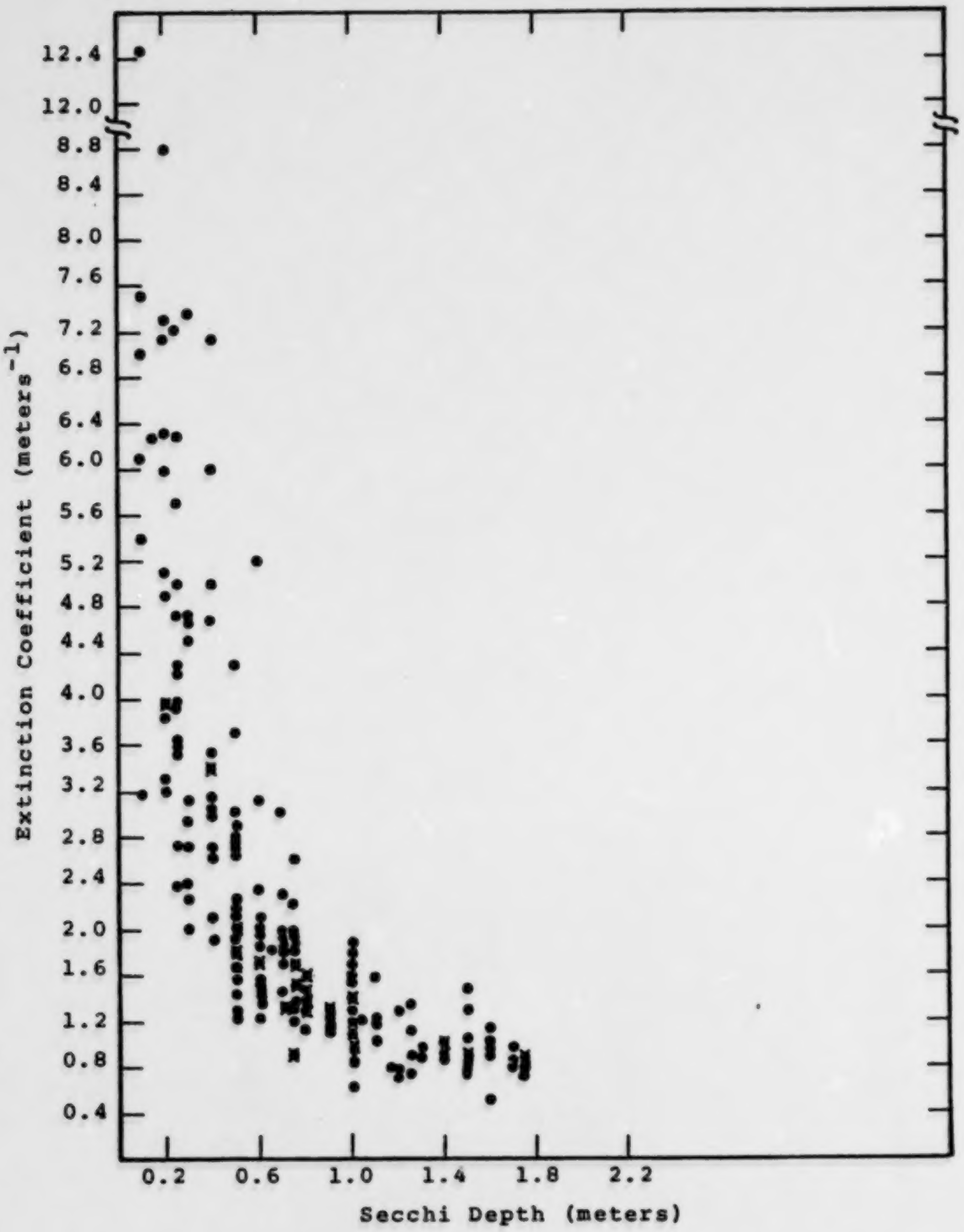


Figure 20. Relationship between Secchi depth and extinction coefficient

The extinction coefficient due to natural turbidity is low in the Main Body. This indicates that the Secchi depth is controlled to a greater extent by the algal population than by inorganic turbidity.

Algae

The algal population of Lake Ray Hubbard can be divided into four major groups: the Cyanophycea (blue-green algae), Chlorophyceae (green algae), Bacillariophyceae (diatoms), and the flagellate group which includes algae from several different classes. The different groups dominate at different times of the year and in different areas of the lake.

In the Main Body, the algal population had a summer and a winter peak as shown in Figure 21. The summer increase in total number of algae, beginning in June during both summers of the study, was mainly due to an increase in the blue-green algal population, particularly Oscillatoria. Other blue-greens which were encountered were Anabaena and Aphanizomenon. Even though the blue-green algae were present in relatively large numbers during late summer, the blue-green algal surface scum frequently associated with nutrient rich waters did not develop. Therefore, one of the most objectionable aspects of the excessive fertilization of a water body, namely the production of the blue-green algal surface scum, is not a problem at this time in Lake Ray Hubbard. The reasons for the absence of the algal scum are unknown. Blue-green algal scums have been observed on other water bodies of the region, so it is possible that this type of problem could develop in Lake Ray Hubbard. Should this happen, there would be additional impetus to control nutrient input to Lake Ray Hubbard because of the extensive recreational use of the lake.

In August 1976, the total number of algae reached 18,000 organisms/ml and during the summer of 1977, the total number reached 19,000 organisms/ml in July. The summer peaks were likely due to the large influx of nutrients to the lake during

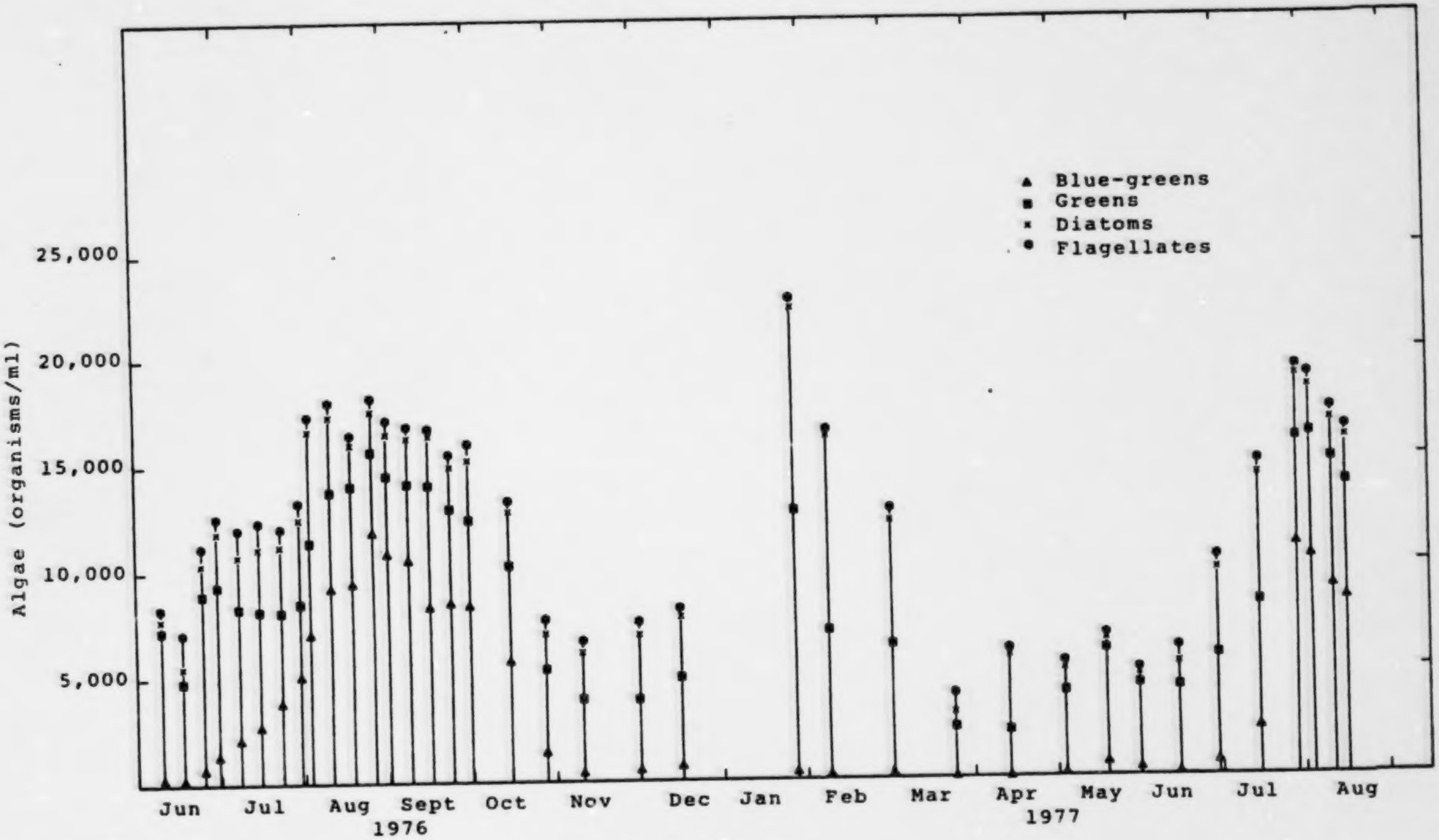


Figure 21. Algal numbers and types - Main Body

the spring combined with the increasing water temperature and light intensity. In June 1976, the concentrations of total N and total P were high with corresponding high concentrations of inorganic N and soluble ortho P, the forms available for algal growth. In 1977, the nutrient concentrations were high during March, April, and May. As the algal numbers increased over the summers, there were corresponding decreases in the nutrient concentrations.

The winter peak of 23,000 organisms/ml occurred in January and was due to large numbers of the green alga, Chlorella. The diatoms Fragillaria and Cyclotella were present but were not as abundant as Chlorella which comprised approximately 60 percent of the population. There was no large influx of nutrients to the lake at this time; however, the lake was isothermal so nutrients were recycled from the sediments and were made available to algae in the surface waters. The algal population in the Main Body was smaller than in the arms of the lake. During the non-peak periods the total number of algae fluctuated around 6,000 organisms/ml.

Chlorophyll a

The concentration of chlorophyll a in a water body is generally indicative of the algal biomass. Samples from each lake station were analyzed for chlorophyll a and its degradation product, pheophytin a. The sum of chlorophyll a and pheophytin a is referred to as uncorrected chlorophyll a.

Figures 22 to 24 are maps of Lake Ray Hubbard showing the uncorrected chborophyll a concentrations throughout the lake on several sampling dates. From these figures, it can be seen that the Main Body had the lowest chlorophyll a concentrations, Rowlett Arm had the highest, and the concentrations in Muddy Creek Arm and the East Fork Arm were generally intermediate between the Main Body and Rowlett Arm. The larger concentrations in the arms of the lake would be expected since most of the nutrients

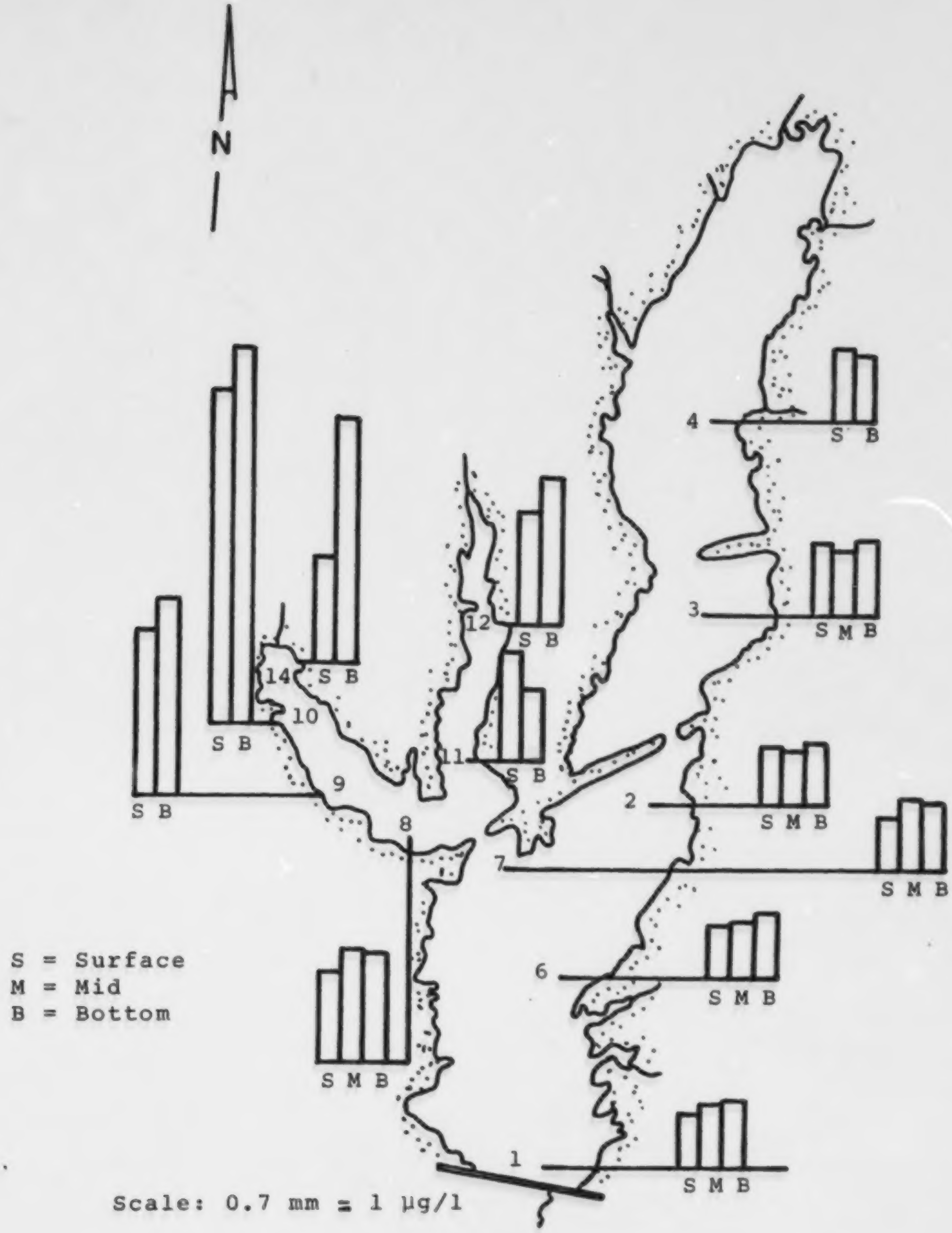

N
4
S B
3
S M B
12
S B
14
S B
10
S B
11
S B
9
S B
2
S M B
8
7
S M B
6
S M B
S = Surface
M = Mid
B = Bottom
S M B
1
S M B
Scale: 0.7 mm = 1 µg/l

Figure 22. Uncorrected chlorophyll a concentration - 9/14/76

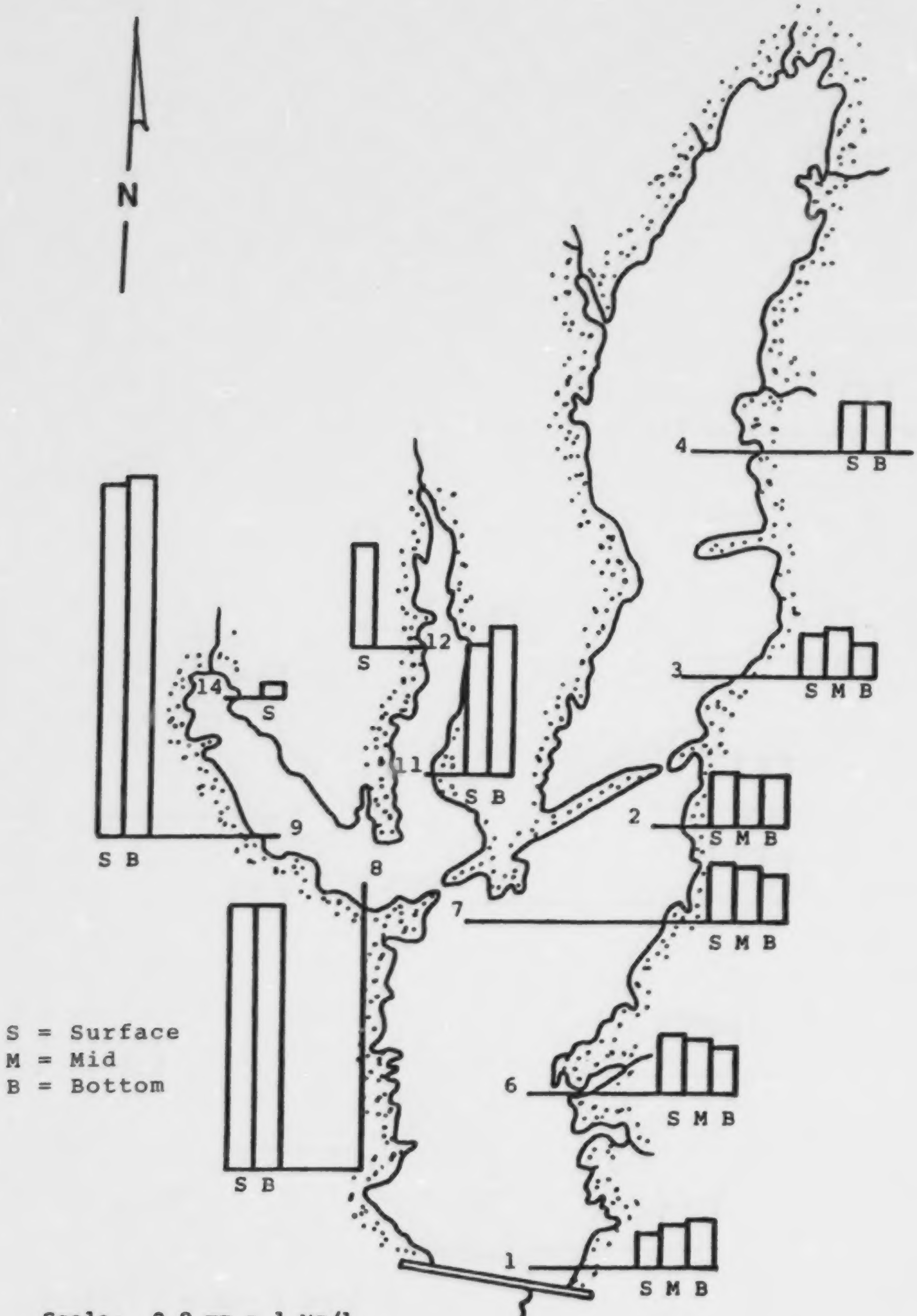
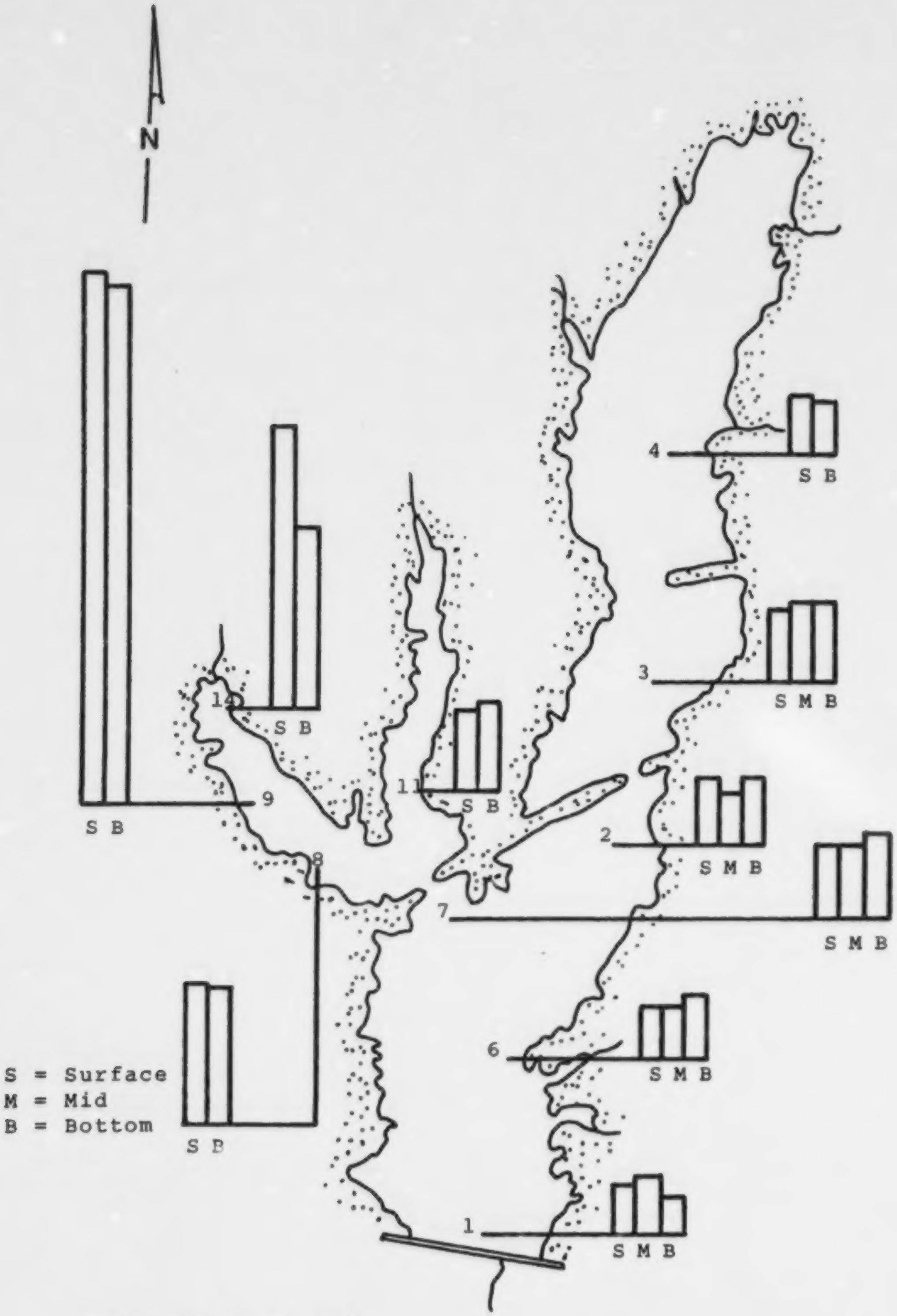

N
4
S B
3
S M B
2
S M B
7
S M B
6
S M B
1
S M B
12
S
11
S B
14
S
9
S B
8
S B
S = Surface
M = Mid
B = Bottom
Scale: 0.8 mm = 1 μg/l

Figure 23. Uncorrected chlorophyll a concentration - 12/14/76


N
4
S B
3
S M B
14
S B
11
S B
9
S B
2
S M B
8
7
S M B
6
S M B
S = Surface
M = Mid
B = Bottom
S B
1
S M B

Scale: 0.8 mm = 1 µg/l

Figure 24. Uncorrected chlorophyll a concentration - 8/9/77

enter the lake from tributaries which empty into the arms. The few small tributaries which discharge directly to the Main Body contribute little to the total nutrient load.

The average concentrations of chlorophyll a and uncorrected chlorophyll a in the Main Body during the study period are shown in Figure 25 . During the periods of thermal stratification (6/8 to 9/7/76 and 3/22 to 8/16/77), the mean of the surface and mid-depth concentrations at each station was plotted. However, during the isothermal period (9/7/76 to 3/22/77), the mean concentration in the water column was plotted. The uncorrected chlorophyll a concentrations ranged from 4 to 20 µg/l during the study period. The concentrations increased during the summer months with peaks of 16 µg/l in October 1976 and 20 µg/l during June 1977. Since the study ended in mid-August 1977, there are no data for the end of the 1977 summer, but there appeared to be an upward trend in August, indicating that the concentrations found during late summer would be high. There was also an increase in the uncorrected chlorophyll a concentration during January 1977 with a peak of 15 µg/l. The chlorophyll a concentrations, ranging from 2 to 12 µg/l, showed the same trend as the uncorrected chlorophyll a concentrations. The chlorophyll a was generally about 60 percent of the uncorrected chlorophyll a, with pheophytin a contributing about 40 percent of the total.

These levels of chlorophyll would place the Main Body of Lake Ray Hubbard in a group with other lakes which are considered moderately eutrophic. Generally, chlorophyll a concentrations above 10 µg/l are associated with eutrophic to highly eutrophic waters. Normally, waters with chlorophyll a concentrations of this magnitude tend to have excessive growths of algae which significantly impair the use of the water for domestic water supplies and recreation.

Figure 26 is a plot of the uncorrected chlorophyll a concentration as a function of the total number of algae. Data from all four areas of the lake are included in this figure. As expected, the relationship between the two is poor. The

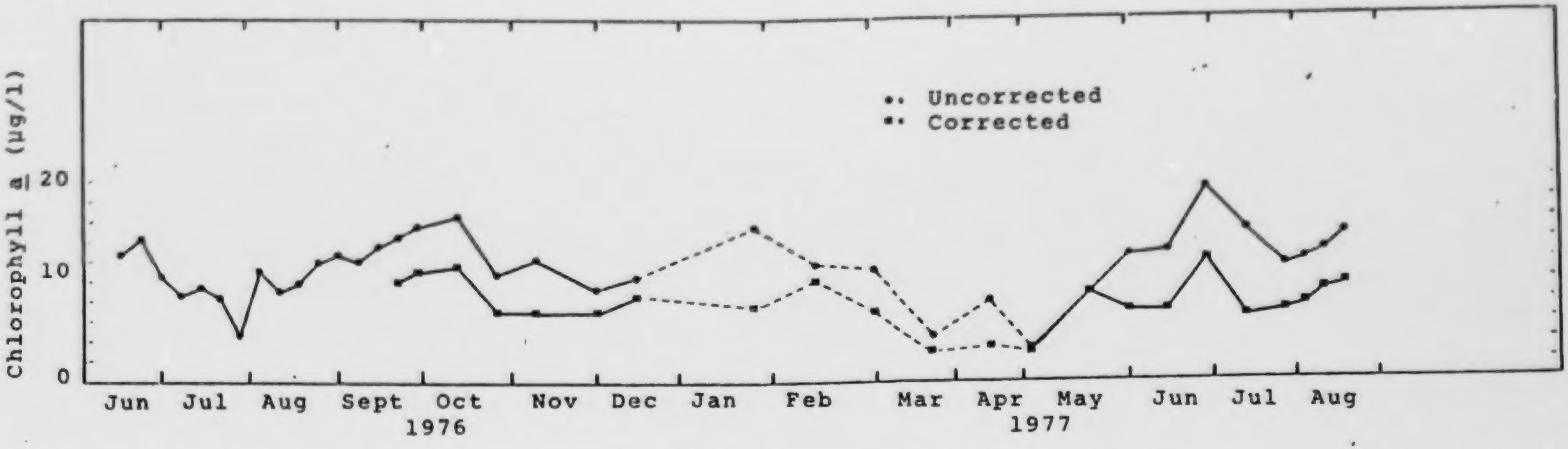


Figure 25. Chlorophyll a concentration - Main Body

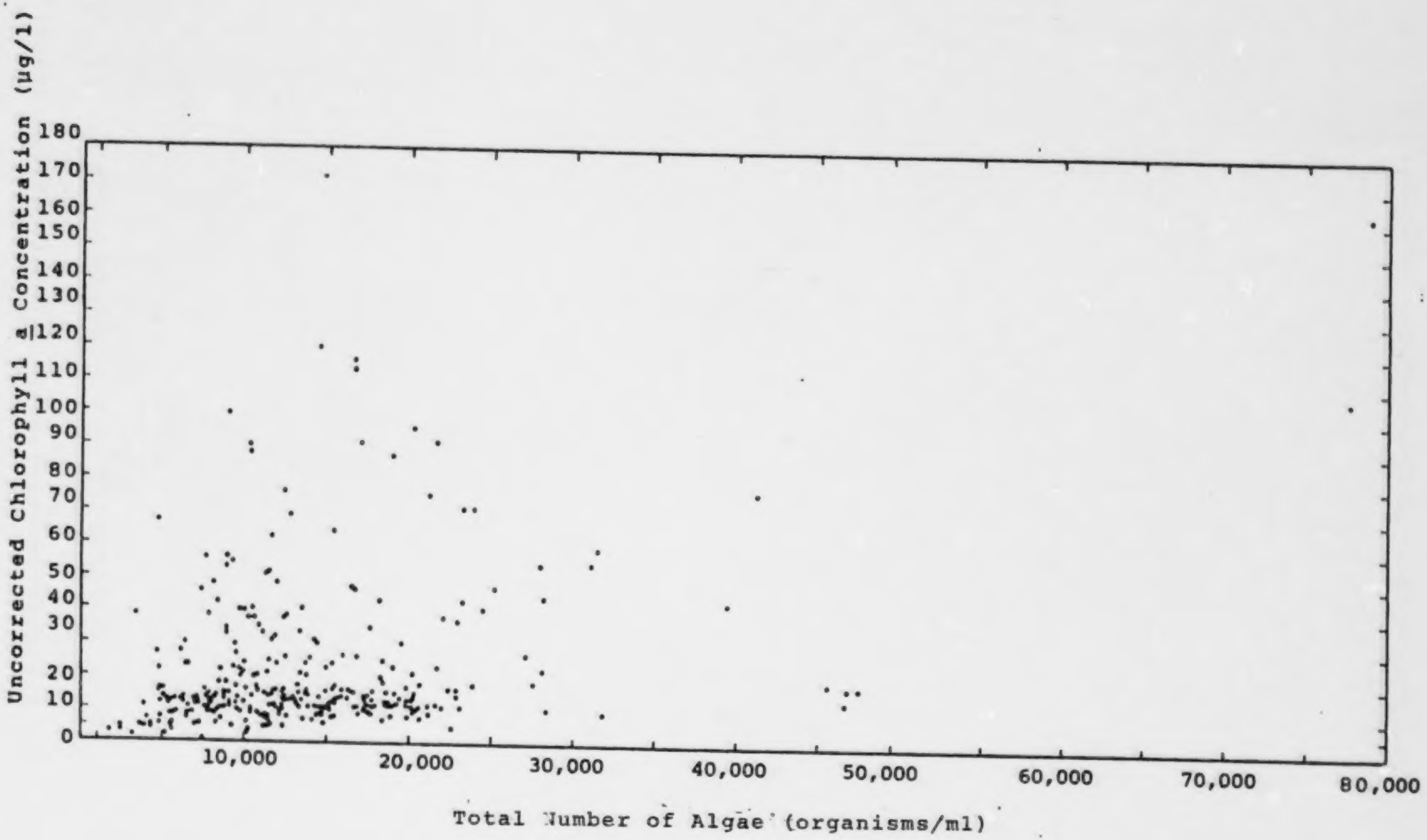


Figure 26. Relationship between total number of algae and uncorrected chlorophyll a concentration.

concentration of chlorophyll a in an algal cell is dependent on many factors including the age and size of the cell. The ease of extraction of chlorophyll depends upon the algal species. Although there was not a good relationship between the total number of algae and the uncorrected chlorophyll a concentration when all data were considered, the plots of uncorrected chlorophyll a and algal numbers showed the same trends over the study period.

Primary Productivity

Table 13 shows the primary productivity measured as mg C fixed/m^2/hr during the late morning and early afternoon in each of the four areas of the lake for the four studies. The depth of the euphotic zone is included for each station and date. The highest primary productivities were found during July and August in the Main Body, and the lowest primary productivities were found during September and March. In general, the highest total production per square meter was found in the Main Body where the euphotic zone varied from 3 m to 5 m, and the primary productivity varied from 266 to 596 mg C/m^2/hr.

Threshold Odor Number

Samples were collected at a depth of 2 m for the determination of the threshold odor number (T.O.N.) and characteristic odor of the water. The average T.O.N. was calculated by taking the mean of the T.O.N.'s at all four of the Main Body stations during the summer months. During the winter months, samples were only collected at Stations 1 and 6 for determination of the T.O.N. so the T.O.N.'s at these two stations were used to calculate the average T.O.N. As Figure 27 shows, the average threshold odor numbers in the Main Body were higher during the summer months, ranging from 6.5 to 13 during 1976 and 6 to 10 during 1977. During the winter months, the T.O.N. was typically near 7. The characteristic odors commonly found were musty, grassy, rooty, fishy, and vegetable.

TABLE 13. RESULTS OF PRIMARY PRODUCTION STUDIES

Region of Lake	July 30, 1976		August 19, 1976		Sept. 23, 1976		March 19, 1977	
	$\frac{\text{mg C}}{\text{m}^2/\text{hr}}$	Euphotic Zone (m)	$\frac{\text{mg C}}{\text{m}^2/\text{hr}}$	Euphotic Zone (m)	$\frac{\text{mg C}}{\text{m}^2/\text{hr}}$	Euphotic Zone (m)	$\frac{\text{mg C}}{\text{m}^2/\text{hr}}$	Euphotic Zone (m)
Main Body	385	5	596	5	266	3.5	319	3
East Fork Arm	353	3	333	2	249	2.5	238	3
Muddy Creek Arm	562	3.5	462	3	128	1.5	234	3
Rowlett Arm	367	2	287	1.5	177	1.5	187	2

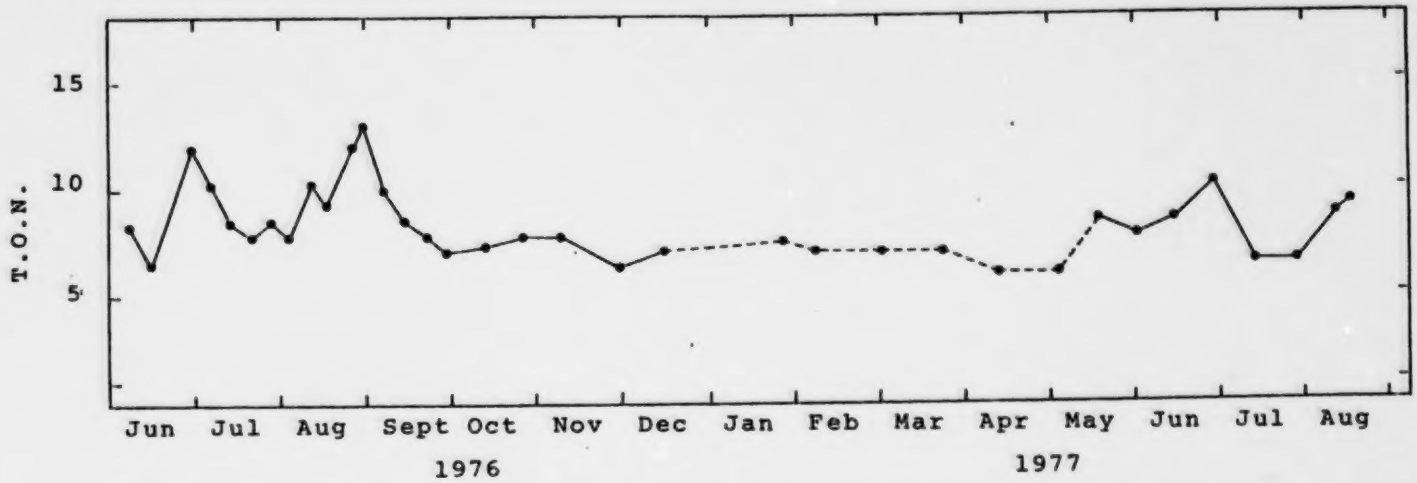


Figure 27. Average threshold odor number - Main Body

It is generally thought that high T.O.N.'s are related to large populations of blue-green algae; however, in Lake Ray Hubbard, major peaks in the T.O.N. were found in early summer when the blue-green algal population was small. Peaks in the T.O.N. in June and July of 1976 and May and June of 1977 corresponded to algal populations which were dominated by the green alga, Chlorella.

Actinomycetes

Starting in February 1977, samples were collected at Stations 1 and 6 in the Main Body for determination of the number of actinomycetes. Actinomycetes are fungus-like bacteria which secrete volatile substances to the water which cause musty and grassy odors (APHA *et al.*, 1976). It is thought that the blue-green algae secrete substances to the water which stimulate the growth of actinomycetes (APHA *et al.*, 1976). There appears to be no relationship between the number of blue-green algae and the average number of actinomycetes. As shown in Figure 28, there were generally 2 actinomycetes/ml, even during the summer months when the blue-green algae and the T.O.N. were increasing.

Currents

Measurements of the direction and magnitude of the currents at Stations 2, 6, and 7 in the Main Body were first made on March 22, 1977. The measurements were made on each sampling trip thereafter at depths of 1, 3, and 5 m. Table 14 presents the current data collected at these stations. In general, the surface currents were flowing to the north with velocities ranging from less than the detection limit to 3.3 cm/sec.

Since the Main Body currents were normally flowing toward the north, water from the Main Body was often flowing into Rowlett Arm, which resulted in mixing of the Rowlett Arm water with its high concentrations of nutrients and the Main Body water with its relatively low nutrient concentrations. From the velocities of the currents, it would be expected that water

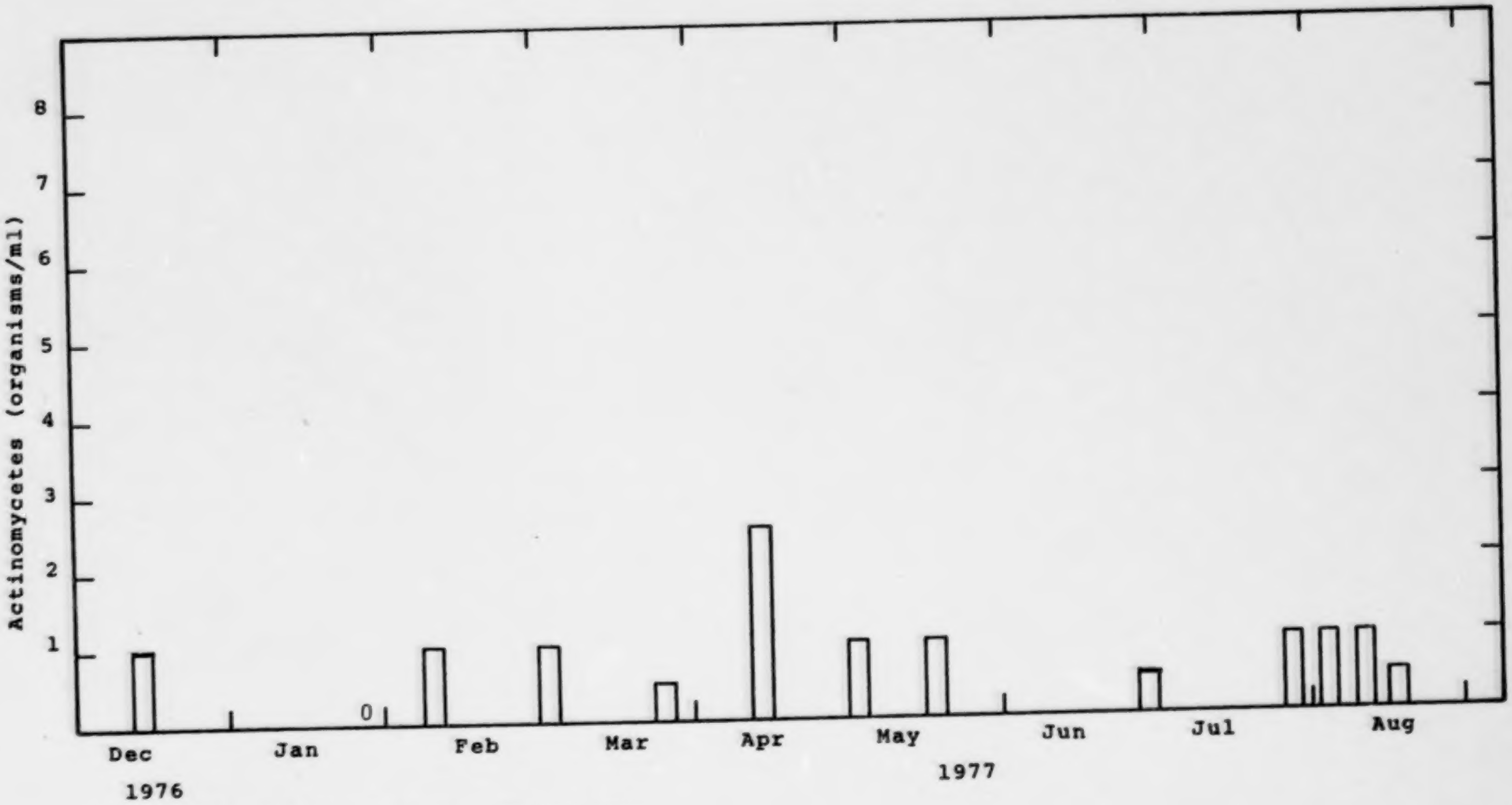


Figure 28 Actinomycetes - Main Body

TABLE 14. LAKE RAY HUBBARD CURRENT MEASUREMENTS- MAIN BODY

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
March 22, 1977				
2	10:25	1	1.9	190
		3	-	-
		5	-	-
6	9:25	1	3.5	150
		3	2.9	170
		5	2.8	190
7	11:20	1	1.7	*
		3	0.7	*
		5	1.1	*
April 12, 1977				
2	10:35	1	-	-
		3	-	-
		5	1.5	140
6	10:10	1	4.0	12
		3	3.7	12
		5	3.0	16
7	9:00	1	1.5	0
		3	0.3	75
		5	0.9	150
May 3, 1977**				
6	9:55	1	2.6	35
		3	1.3	0
		5	2.5	340
7	9:00	1	3.6	30
		3	2.0	20
		5	1.6	340

(Continued)

Table 14 (Continued)

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
May 17, 1977				
2	10:00	1	2.5	335
		3	2.7	10
		5	2.6	10
6	9:47	1	3.6	350
		3	2.8	335
		5	-	-
7	8:47	1	-	-
		3	8.7	160
		5	2.6	340
May 31, 1977				
2	11:32	1	4.4	205
		3	4.1	200
		5	-	-
6	10:56	1	2.6	220
		3	0.8	205
		5	3.3	210
7	8:40	1	2.9	120
		3	1.5	230
		5	2.6	180
June 14, 1977				
7A	8:36	1	-	-
		3	-	-
		5	2.5	110
7	8:57	1	1.2	250
		3	-	-
		5	1	250

(Continued)

Table 14 (Continued)

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
6	10:15	1	2.3	240
		3	1.3	230
		5	-	-
2	10:53	1	2.6	210
		3	1.6	210
		5	2.0	160
June 28, 1977				
7	13:27	1	5.9	5
		3	5.4	350
		5	1.2	70
6	10:26	1	2.6	0
		3	4.1	15
		5	1.1	0
2	12:00	1	3.2	20
		3	2.4	30
		5	2.9	50
July 12, 1977				
2	10:45	1	-	-
		3	-	-
		5	-	-
6	9:55	1	1.5	330
		3	1.2	355
		5	2.4	340
7	9:00	1	2.3	0
		3	3.0	15
		5	-	-
7A	8:30	1	-	-
		3	-	-
		5	-	-

(Continued)

Table 14 (Concluded)

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
July 26, 1977				
2	10:50	1	-	-
		3	-	-
		5	2.6	0
6	10:15	1	0.5	340
		3	-	-
		5	-	-
7	9:25	1	1.3	340
		3	-	-
		5	-	-
August 2, 1977				
2	10:30	1	1.0	*
6	10:10	1	2.5	160
7	9:00	1	0.8	160
7A	8:50	1	-	-
August 9, 1977				
2	10:30	1	3.5	0
6	10:05	1	2.7	0
7	8:50	1	2.5	25
7A	8:25	1	2.9	17
August 16, 1977				
2	11:20	1	4.6	*
6	10:40	1	3.6	0
7	9:25	1	2.8	0
7A	8:40	1	1.2	340

Dash (-) indicates no measurable current.
*Indicates not measured.
**Other stations were not sampled because of breakdown of DWU boat.

entering the Main Body would be well mixed horizontally in a short period of time. At an average velocity of 3 cm/sec, it would take 2.5 days for water to flow from the dam to Station 7. This explains why the Main Body of the lake was found to be horizontally homogeneous with respect to the Cl^- content even though the Cl^- enters the lake primarily from Rowlett Arm.

Figure 29 illustrates typical hourly wind speeds at the dam. The graph clearly shows that there is a daily cycle with the highest winds in the afternoon and lowest winds during the evening and early morning hours. There may also be short periods of high wind associated with thundershower activity during the summer months. The surface current velocities in the lake would be expected to be greater in the afternoon due to the higher afternoon wind velocities.

Figures 30 and 31 show the percent of time the wind was blowing from each direction and the average wind speed from each direction during the four seasons of the year. During the spring months, the wind blew from the southeast or south over 60 percent of the time. The average wind velocity was approximately 16 km/hr regardless of the direction from which it was blowing. During the summer months, the wind blew from the south or southeast approximately 75 percent of the time with average velocities ranging from 8 to 16 km/hr. The wind blew from the southeast about 40 percent of the time at average velocities of 8 to 16 km/hr during the fall months. About 35 percent of the time the wind was blowing from the north or northwest at average velocities of 16 to 30 km/hr. During the winter months, the wind blew from the south-southeast 40 percent of the time at average velocities of 8 to 20 km/hr. The wind blew from the north 15 percent of the time at about 25 km/hr and from the northwest 10 percent of the time at average velocities of 10 to 25 km/hr. As shown in the figures, the wind blows mainly from the south throughout the year. This indicates that water would be flowing from the Main Body up into the arms of the lake during most of the time that the wind was blowing.

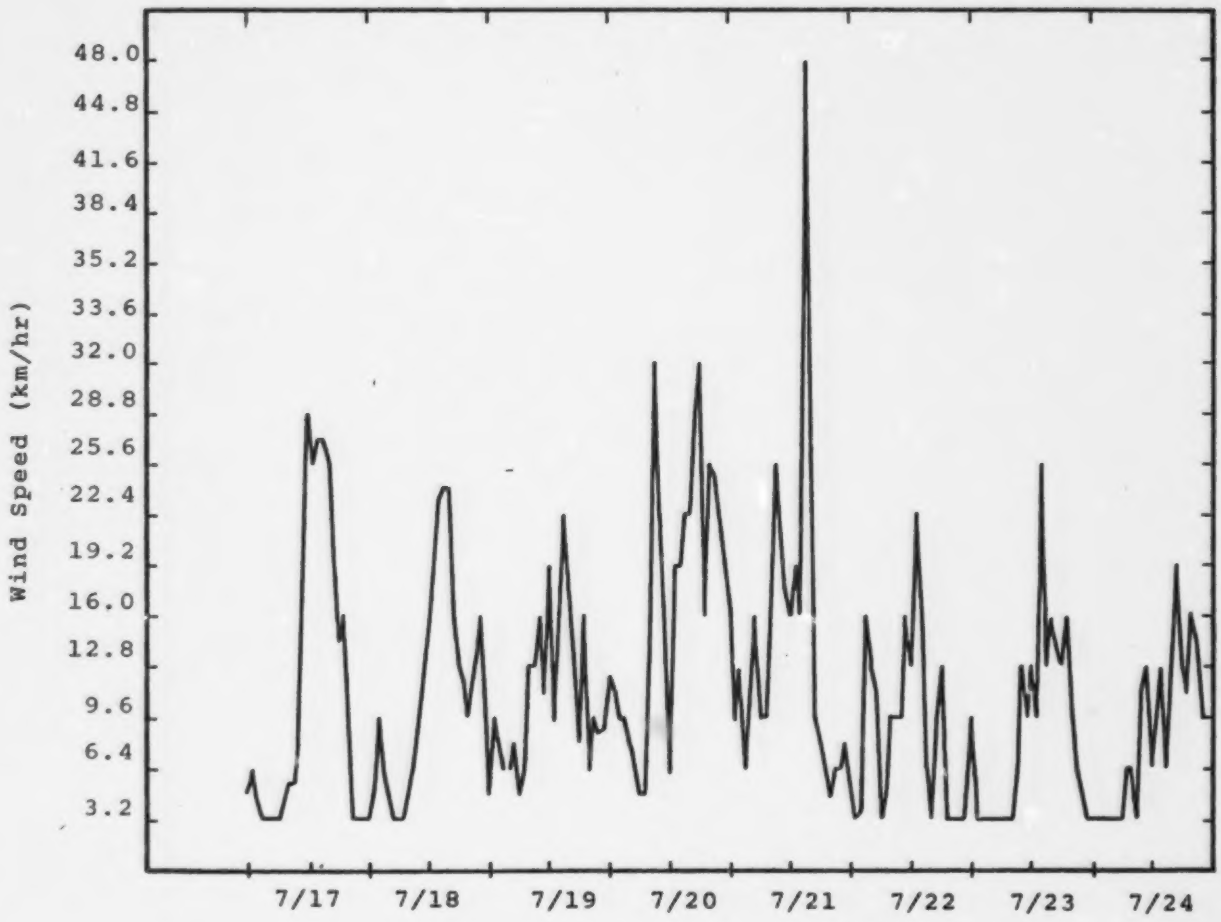


Figure 29. Hourly wind speeds - July 17-July 24, 1977

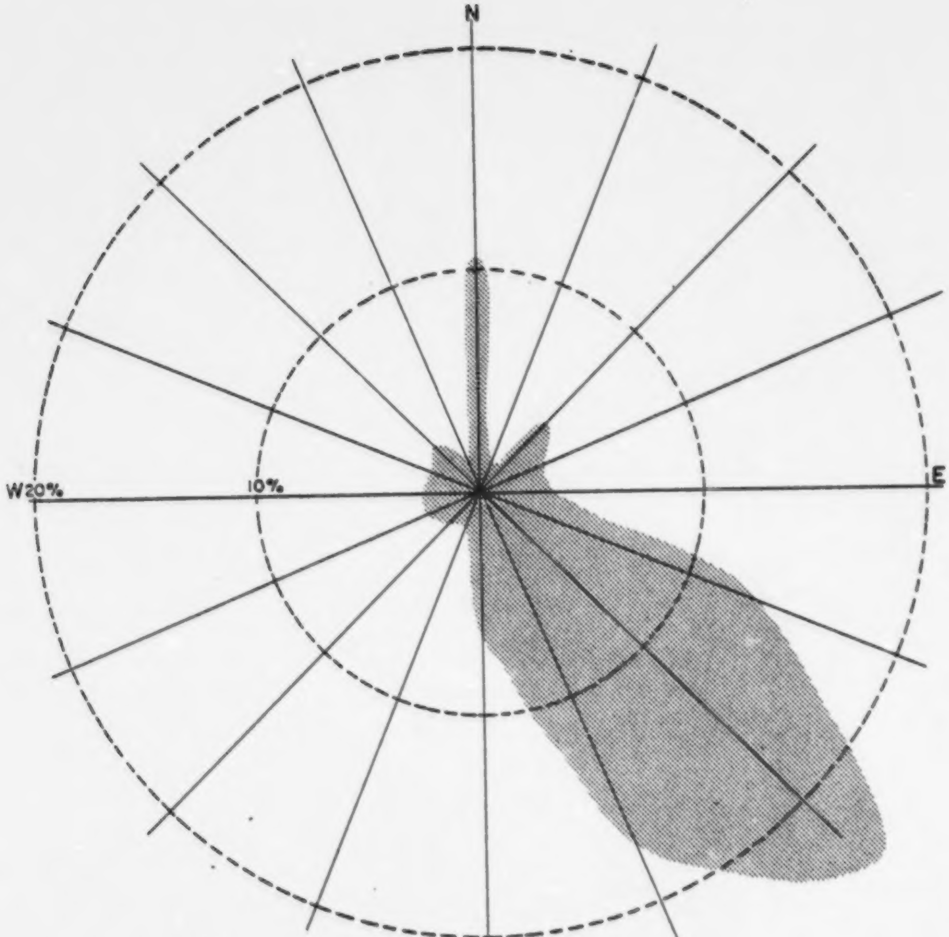


Figure 30a. Percent of time wind blew from each direction during spring

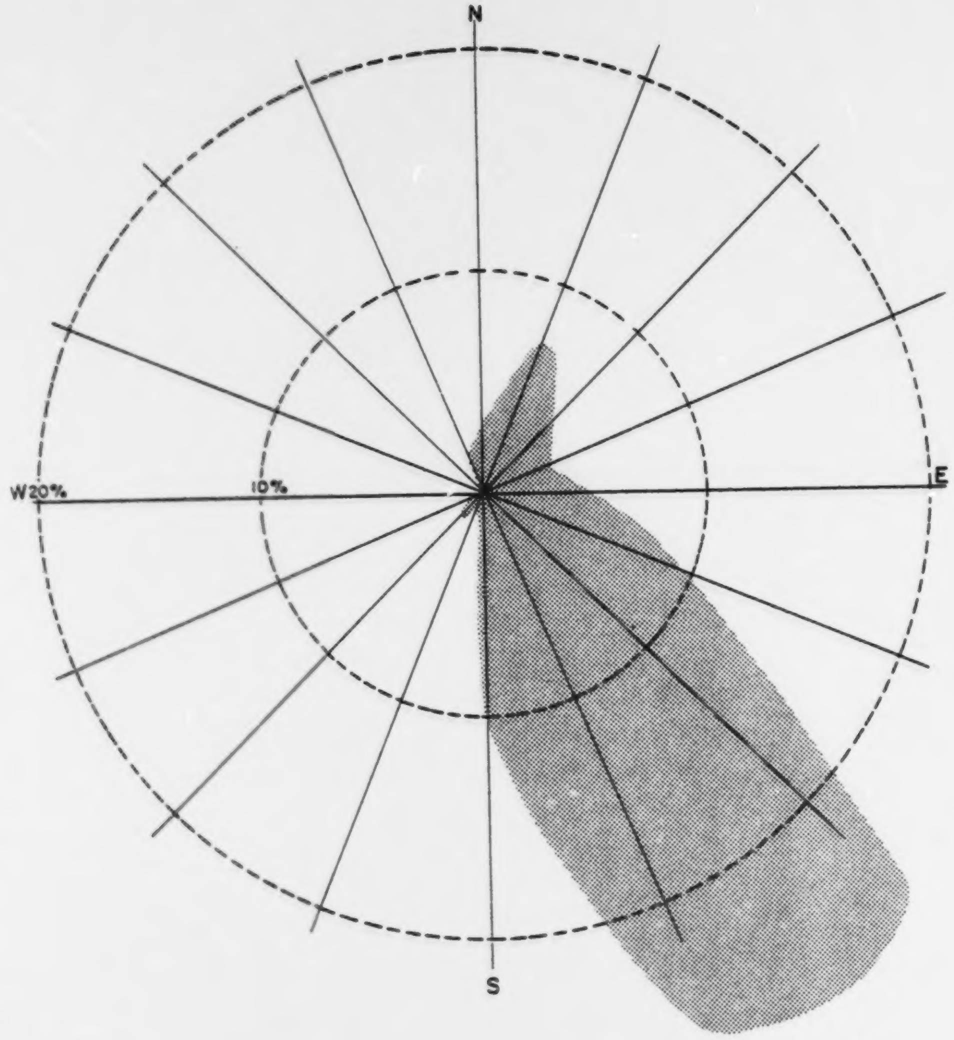


Figure 30 b. Percent of time wind blew from each direction during summer

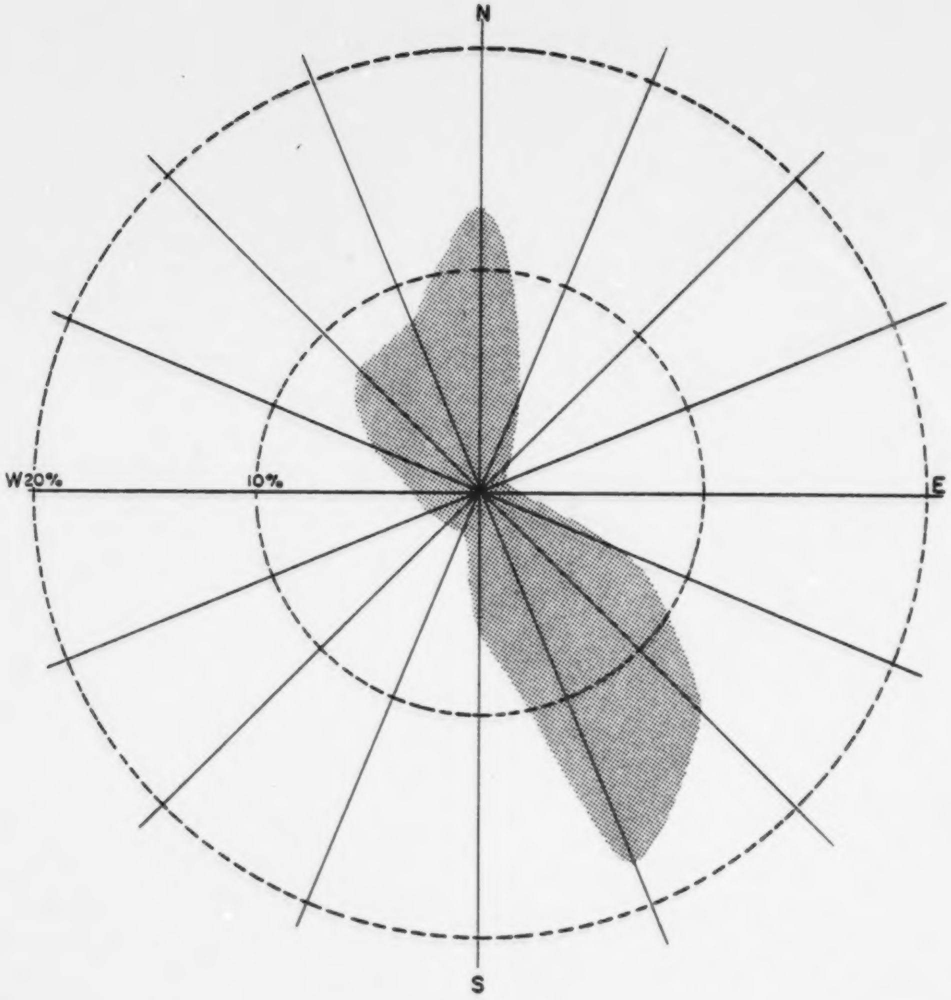


Figure 30c. Percent of time wind blew from each direction during fall

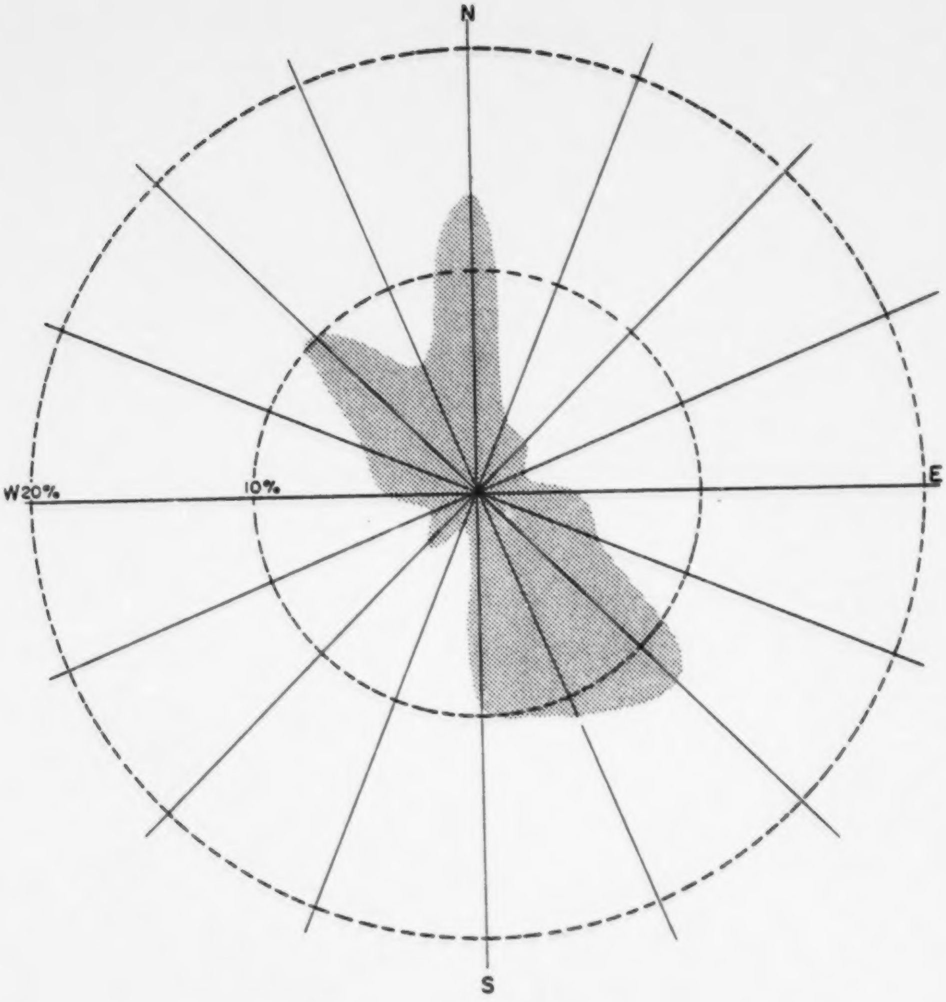


Figure 30d. Percent of time wind blew from each direction during winter

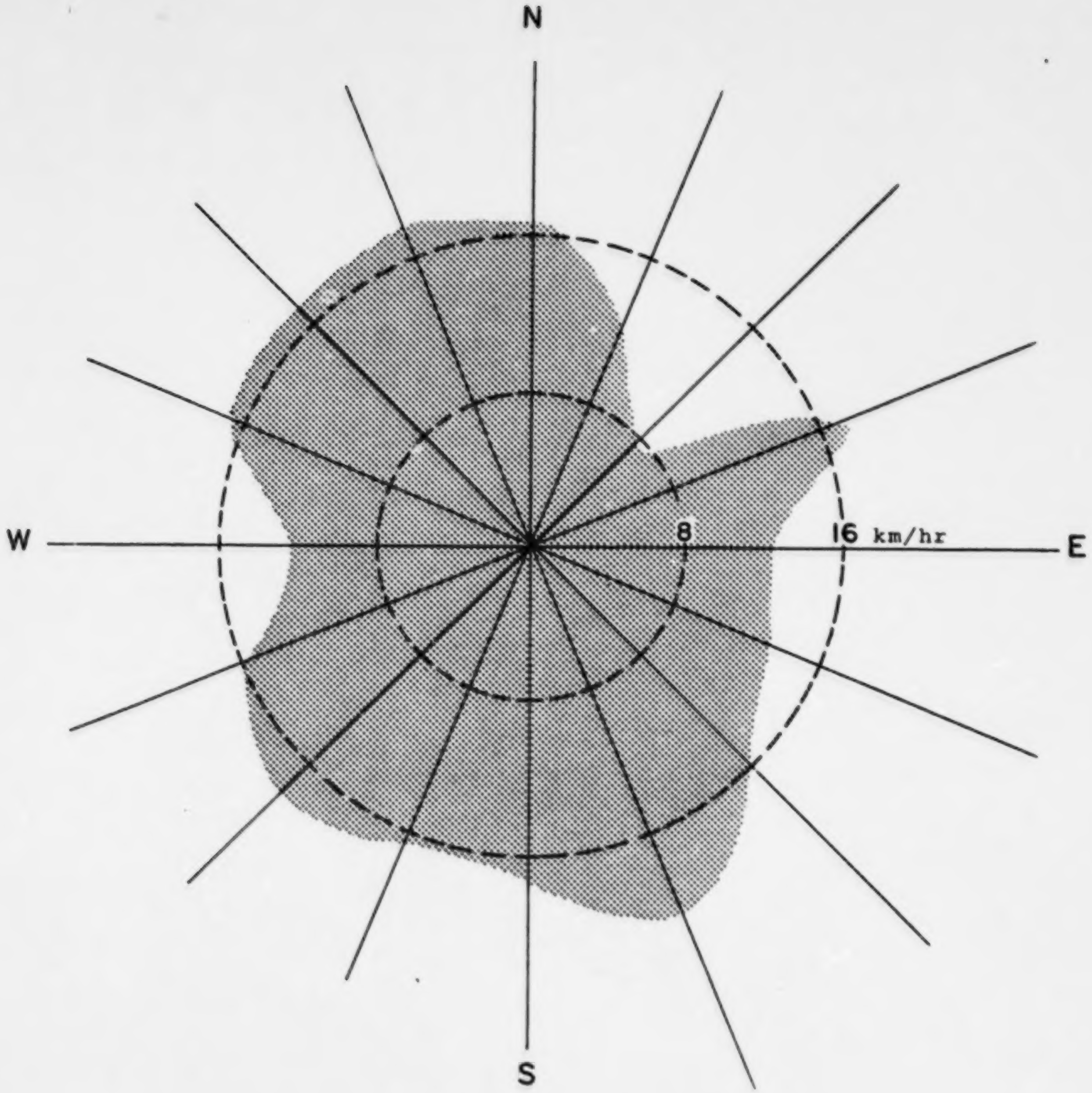


Figure 31a. Average wind speed from each direction during spring

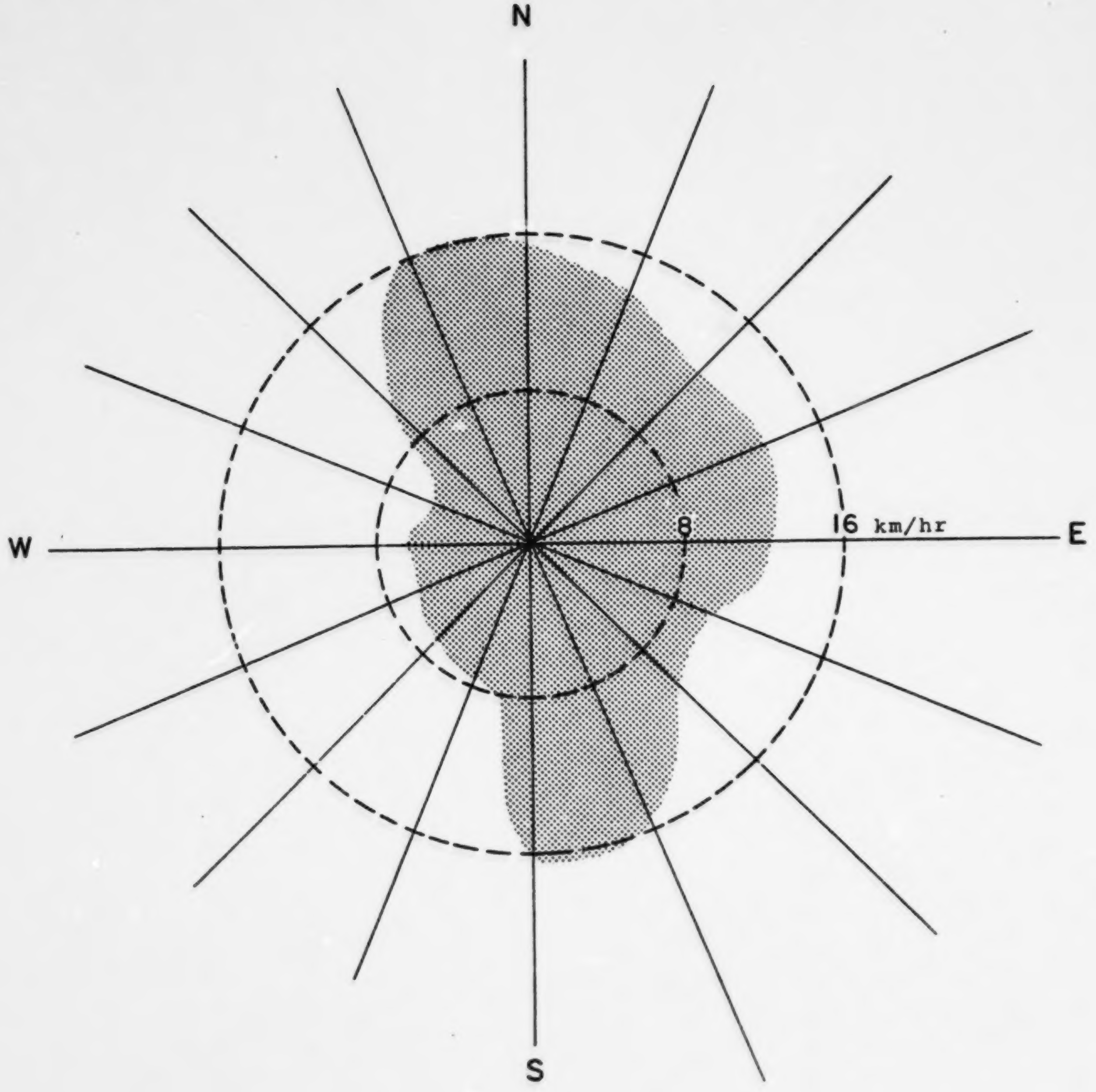


Figure 31b. Average wind speed from each direction during summer

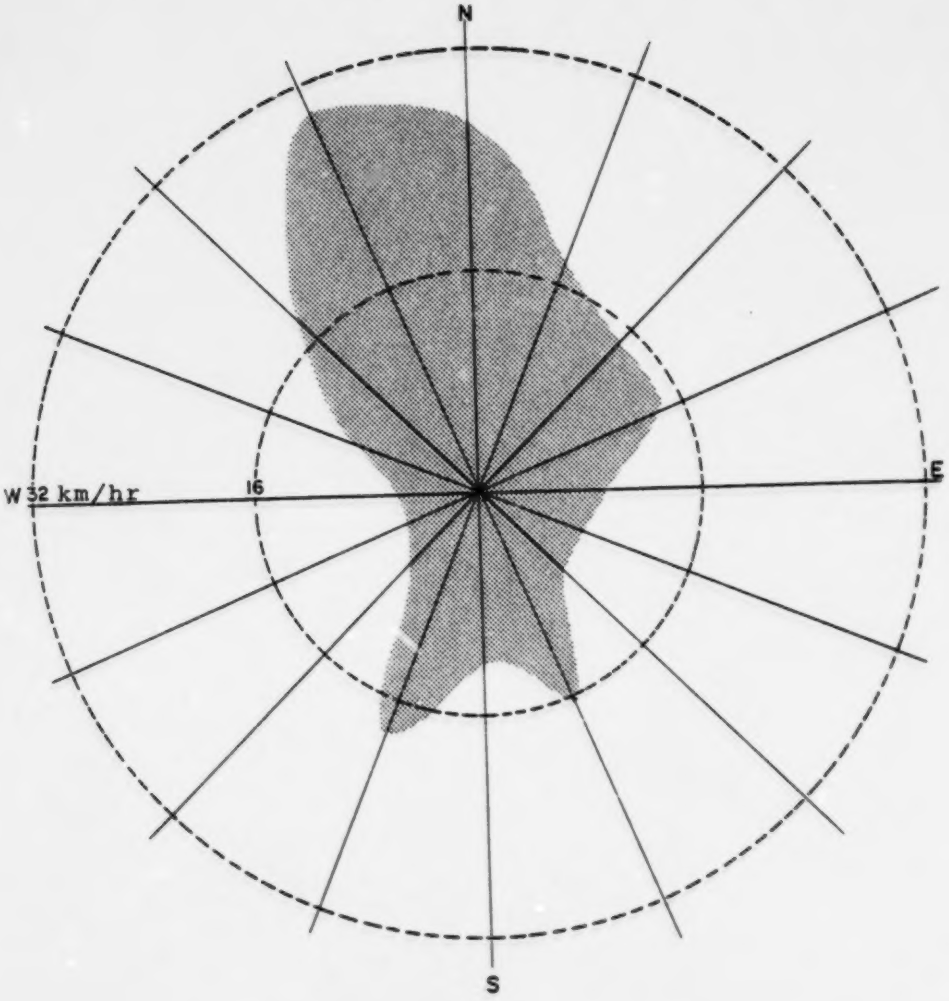


Figure 31c. Average wind speed from each direction during fall

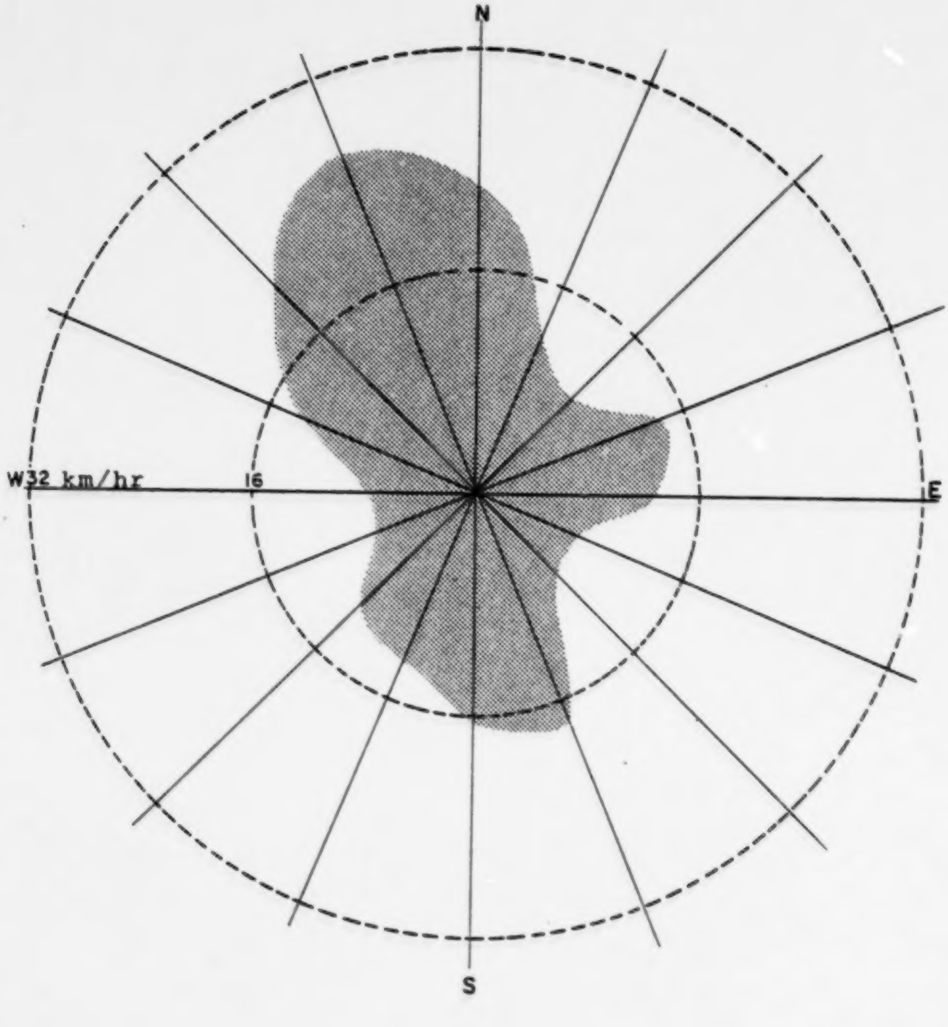


Figure 31d. Average wind speed from each direction during winter

It is expected that the daily water level fluctuations are such that a substantial flow occurs between the Main Body and the arms. This is supported by the chloride data which show that water entering Rowlett Arm from Rowlett Creek was diluted 1 to 10 by the time it reached Station 8. Attempts were made to see if the water level recorder at the dam reflected the periodic change in elevation due to the wind. No detectable change could be seen with the equipment at this location, although such changes would likely have been detected if more sensitive equipment had been used.

From the chemical data on the lake, it appears that the wind-induced currents were sufficient to suspend considerable amounts of sediment. This was particularly evident during the fall, winter, and spring months when phytoplankton numbers were low and the water clarity was also low. The thermocline in the Main Body of the lake would be expected to play a significant role in preventing the suspension of sediment from the bottom during the summer in those areas of the lake that are greater than 10 m deep and, therefore, have water below the permanent thermocline. Since only 42.5 percent of the area of the Main Body has a depth of 10 m or greater and, therefore, is under permanent stratification during the summer, it is somewhat surprising that the thermocline has such a pronounced effect on the suspension of sediments throughout the Main Body. Even though there was not permanent stratification in the water column in areas having a depth less than 10 m, at many times, temporary stratification was found at Stations 2, 6, and 7. This temporary stratification appeared to play a significant role during the summer in preventing suspension of sediments.

As indicated in the modeling section, it was necessary to estimate the nutrient input to the Main Body of Lake Ray Hubbard based on the concentrations of nutrients at Stations 3, 8, and 11 and the tributary flows to each of the respective parts of the lake. It would be expected, because of the oscillatory wind pattern, that the actual flow from the arms to the

Main Body would be considerably greater than the flow from the tributaries. During the morning and especially the afternoon, the wind should drive water up into each of the arms from the Main Body. At night, this water, plus the tributary inflow water, enters the Main Body. In a sense, the arms are acting like small estuaries with daily tidal flows.

CHARACTERISTICS OF ROWLETT ARM

Dissolved Oxygen and Temperature

Rowlett Arm was usually completely mixed with relatively constant temperatures throughout the water column. However, there was normally a decrease in the DO concentration near the bottom, most likely due to the sediment oxygen demand and the bacterial decomposition of the large quantity of organic matter settling from the surface waters. Figure 32 shows several DO and temperature profiles at Station 9 in Rowlett Arm. The surface waters were normally supersaturated or just at saturation with respect to DO. This is most likely related to the photosynthetic production of oxygen by the large algal population.

As expected, there was no permanent thermal stratification of the water column in Rowlett Arm during the summer months as there was in portions of the Main Body. Surface waters were generally about 2°C higher than in the Main Body as shown in Figure 8 . The DO was never completely depleted in the bottom waters of Stations 8 and 9; however, there was DO depletion in the bottom waters at Station 14 during the summer months.

Phosphorus

The average total P concentrations in Rowlett Arm are shown in Figure 33 , and the average soluble ortho P concentrations are shown in Figure 34 . The concentrations of both fluctuated erratically and did not appear to be related to the fluctuations in the algal numbers. Since this arm of the lake had a maximum depth of 5 m with a large section less than 2 m deep, wind and organism-fish induced suspension of the sediments accompanied by

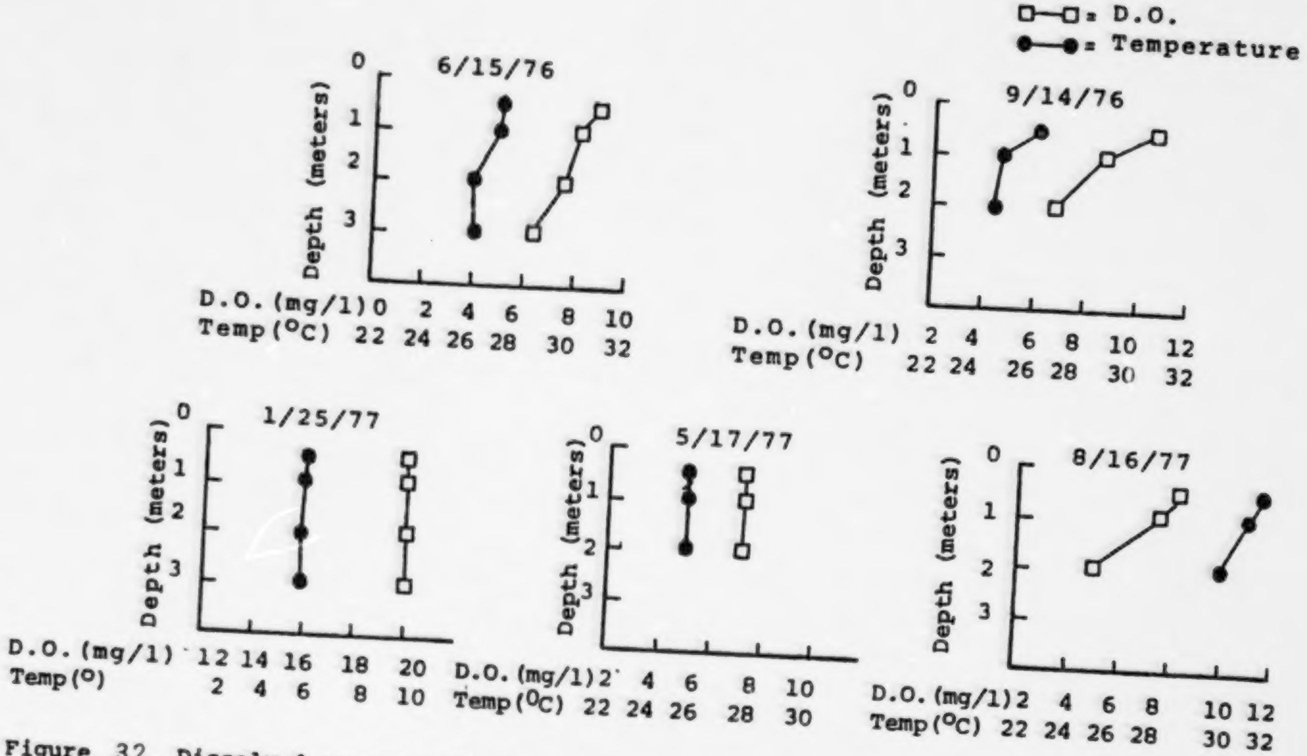


Figure 32. Dissolved oxygen and temperature profiles - Station 9

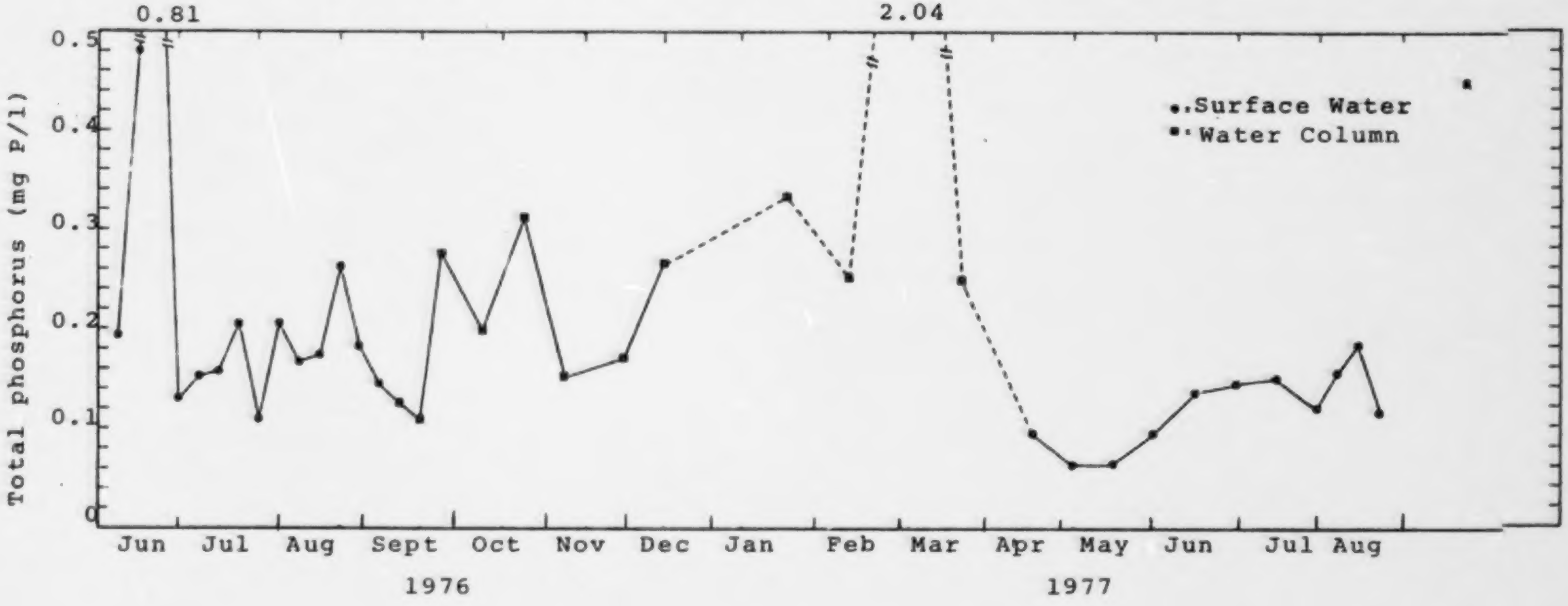


Figure 33. Average total phosphorus concentration - Rowlett Arm

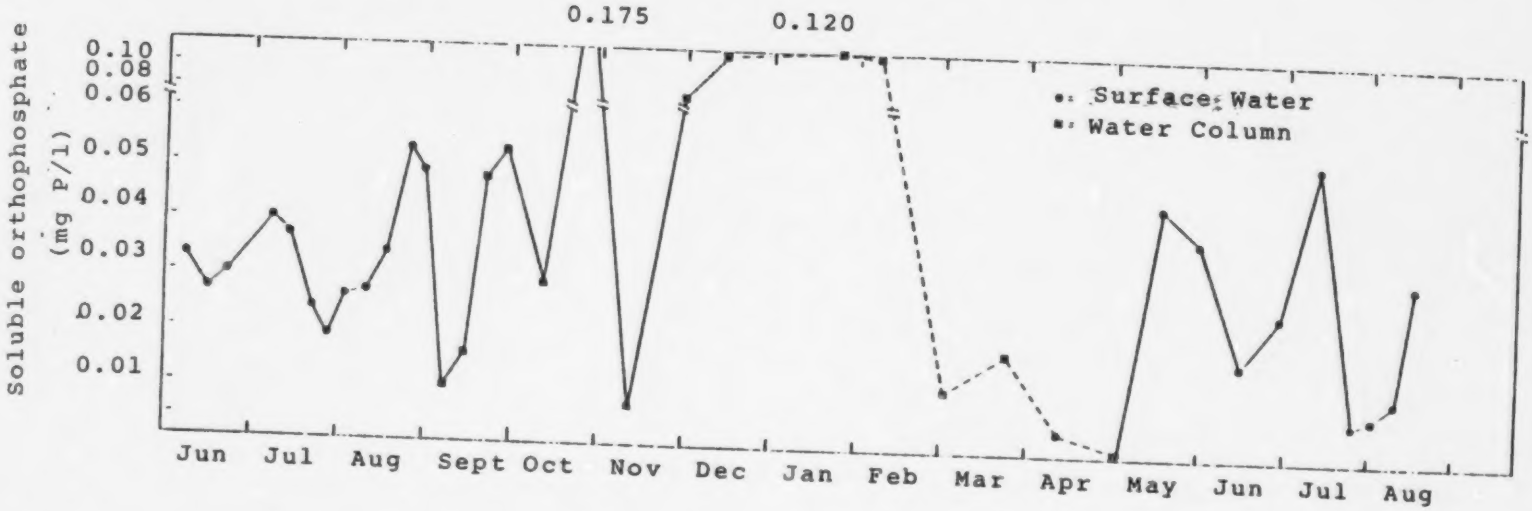


Figure 34 . Average soluble orthophosphate concentration - Rowlett Arm release or uptake of phosphorus by the suspended particles could explain the highly erratic concentrations of total P and soluble ortho P found.

The total P concentrations in Rowlett Arm were high compared to concentrations normally found in the Main Body of Lake Ray Hubbard. As shown in Figure 33, the total P concentration normally ranged from 0.06 to 0.32 mg-P/l with peaks of 0.81 mg-P/l in June 1976 and 2.04 mg-P/l in March 1977. The soluble ortho P concentrations ranged from 0.001 to 0.05 mg-P/l throughout most of the year; however, there was a peak of 0.175 mg-P/l at the end of October 1976. From the end of December 1976 to February 1977, the concentrations ranged from 0.08 to 0.12 mg-P/l. This corresponded to a period of high total P concentration.

Effluent from the Garland and Plano sewage treatment plants is discharged into Rowlett Creek, the tributary to Rowlett Arm. Since domestic wastewater usually has high concentrations (5-10 mg-P/l) of phosphorus relative to most natural waters, the high concentrations found in Rowlett Arm are readily explainable. During low flow conditions, the Rowlett Creek flows are derived primarily from sewage treatment plant discharges. Much of the phosphorus entering Rowlett Arm is removed from the water column in the arm since the total P concentrations in the Main Body are much lower than those in Rowlett Arm. The high concentrations of P found in the upper end of Rowlett Arm at Station 14 compared to the lower concentrations found in the lower end of the arm at Station 8 show that significant P removal occurs in Rowlett Arm.

Nitrogen

Figure 35 shows the average total nitrogen (nitrate, nitrite, ammonia, and organic N) concentrations in Rowlett Arm during the study period. The concentrations of total N fluctuated between 1.0 and 3.9 mg-N/l with the highest concentrations found during June and October to November 1976, and January to May 1977. The June 1976 peak was most likely due to the large

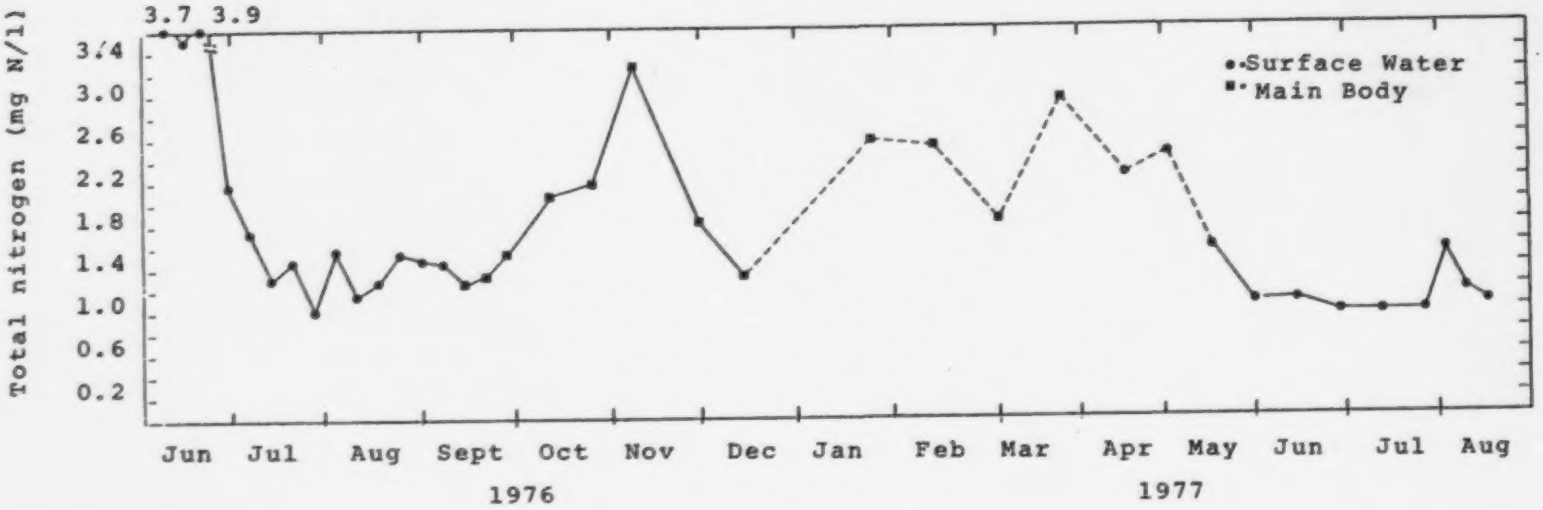


Figure 35. Average total nitrogen concentration - Rowlett Arm

input of nutrients from Rowlett Creek during the spring. The high concentrations found during October and November 1976 and January 1977, were likely due to recycling from the sediments since Rowlett Creek contributed less nitrogen at this time compared to the spring months because the flows were generally relatively low during the fall and winter months. From February to March, a large amount of nitrogen entered Rowlett Arm from Rowlett Creek (see discussion of nutrient loads, Table 23). This explains the high concentrations of total N found in Rowlett Arm during the spring months. During non-peak periods, the total N was typically near 1.2 mg-N/l, which was slightly higher than the non-peak concentrations found in the Main Body. This is likely due to the inputs of nitrogen from the sewage treatment plant and land runoff which enter Rowlett Creek.

The average inorganic N (nitrate, nitrite, and ammonia) concentrations in Rowlett Arm are shown in Figure 36 . The inorganic N concentration was less than 0.1 mg-N/l during most of the summer of 1976 and the summer of 1977. These low concentrations were found during the periods of intense algal blooms in Rowlett Arm. There was a peak of 0.95 mg-N/l during October 1976, likely due to recycling of N from the sediments shortly after the lake became isothermal. In January 1977, the inorganic N concentration started to increase and reached a peak of 2.01 mg-N/l during May. During February, March, and April 1977, large amounts of N entered the lake from Rowlett Creek (see discussion of nutrient loads). During May, the algal population started to increase with the inorganic N being taken up by the algae.

Nitrogen to Phosphorus Ratio

Figure 37 shows the N:P mass ratios in Rowlett Arm. From the beginning of the study in June 1976 until February 1977, the ratios were typically less than 7, indicating that the nutrient likely limiting algal growth in Rowlett Arm was nitrogen during that time. This is to be expected since large amounts of

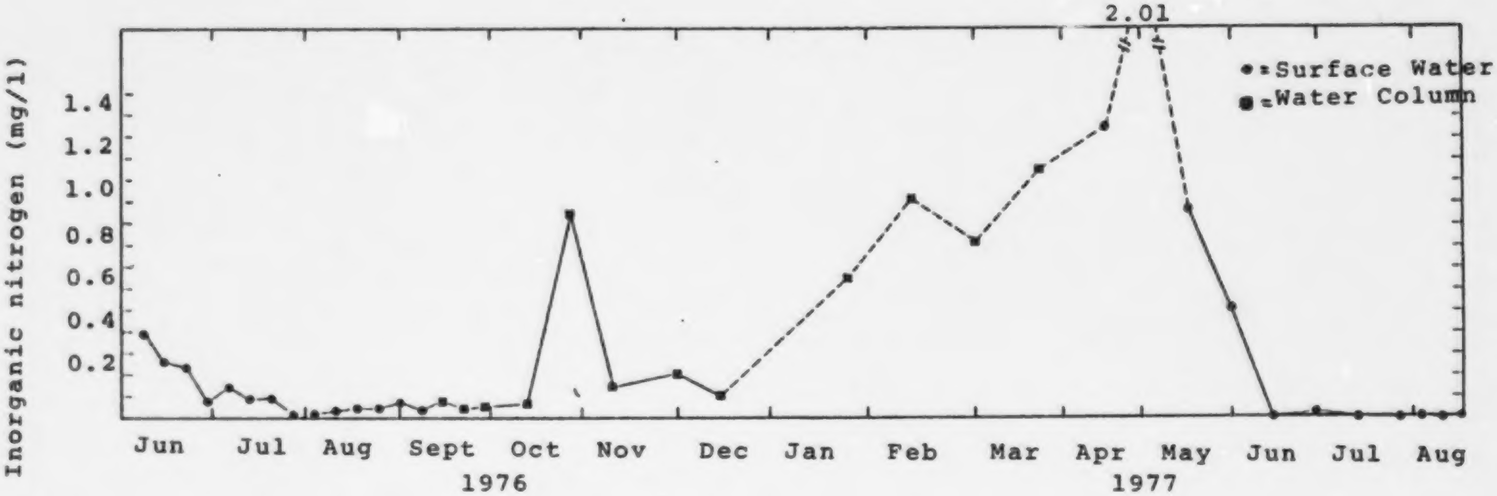


Figure 36. Average inorganic nitrogen concentration - Rowlett Arm

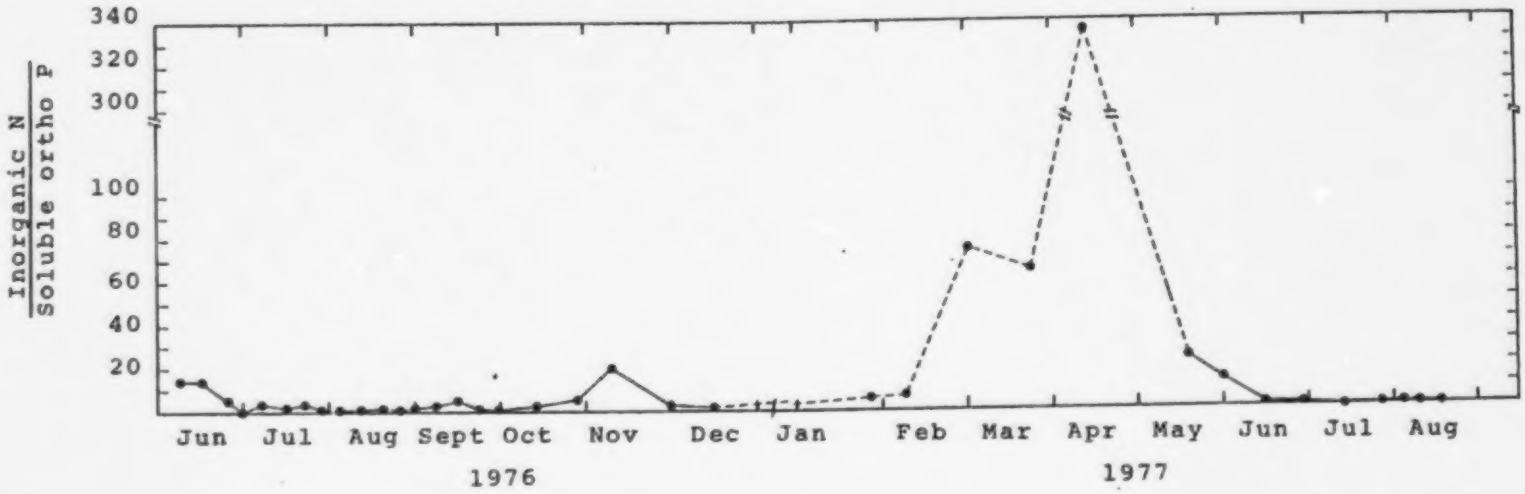


Figure 37. Inorganic nitrogen to soluble orthophosphate ratios - Rowlett Arm

phosphorus entered Rowlett Creek from the Garland and Plano sewage treatment plants. During the spring months, large amounts of N entered Rowlett Arm due to runoff from the land associated with the high spring flows. Based on the N:P ratios, it would appear that there was excessive N during the spring months; however, the actual concentrations of inorganic N and soluble ortho P were sufficiently high to prevent either element from limiting algal growth. During May, the algal population started increasing, decreasing the inorganic N concentration.

During the summer of 1977, the N:P ratios were less than 7 indicating the potential for nitrogen limitation. There was limited land runoff at this time so the phosphorus from the sewage treatment plants was causing Rowlett Arm to be nitrogen limited with respect to algal growth.

Nitrogen Fixation

Nitrogen fixation is a potential source of N for Lake Ray Hubbard which was not studied in this investigation. This phenomenon involves the conversion of dissolved nitrogen gas to algal available forms of N. In natural waters, nitrogen fixation is associated with blue-green algae and bacteria. Generally, bacterial rates of N fixation in aquatic systems appear to be relatively slow. Rates of fixation by blue-green algae, however, may be much more rapid. Within those blue-greens which have the potential ability to fix N, only those possessing heterocysts have been found to actually fix nitrogen. Torrey and Lee (1976) and Torrey (1972) have presented a comprehensive review of the factors influencing N fixation in natural waters and the results of a detailed study of this phenomenon in Lake Mendota near Madison, Wisconsin. For further information on the general aspects of these topics, consult these references.

The potential for N fixation in Rowlett Arm could lead to increased algal growth in this part of the lake as a result of the apparent N limitation of algae in these waters. Nitrogen fixation may be contributing to algal growth in the arm during late summer. N fixation could also be occurring in the Main Body and other parts of the lake where blue-green algae have been found to be dominant. Microscopic examination of some samples has shown that some blue-greens in Lake Ray Hubbard possess heterocysts and, therefore, have the potential to fix nitrogen.

pH

In Rowlett Arm, the average pH was, in general, higher and more variable than in the other areas of the lake. As shown in Table 15, the pH ranged from 7.5 to 8.4 during the summer of 1976 with the highest values found at Stations 9 and 10, most likely due to the uptake of CO_2 by the large algal populations in this area of the lake. During the winter months, the pH ranged from 7.8 to 8.4 with the lowest values found at Station 14 which had a small population of algae. In the summer of 1977, the pH values ranged from 7.8 to 8.9 and, in general, were higher than the pH values found during the previous summer. The pH at Station 9 was highest, corresponding to the high algal population, whereas at Station 8, the pH was only slightly higher than the pH of the Main Body at Station 7.

Figure 38 shows the pH in the surface and bottom waters at Station 9 during the study period. Unlike the pH of the Main Body, the pH values in the surface and bottom waters in Rowlett Arm were essentially the same throughout the year. Since Rowlett Arm is shallow and exposed to the wind, the water column was essentially isothermal throughout the year. There was, therefore, limited restriction to mixing between surface and bottom waters. During the summer of 1976, the pH ranged from about 8 to 9 which was higher than the range of pH values found in the Main Body. This difference was most likely due to the photosynthetic uptake of CO_2 by the large algal populations in Rowlett Arm. The pH increased during the winter months when it ranged from 8.5 to 10.3. The increase in the pH corresponded to an increase in the specific conductance, total

TABLE 15. AVERAGE pH - ROWLETT ARM

Depth	Station 8	 9	 10	 14
6/76 to 9/76				
S	8.1	8.3	8.3	7.6
B	8.0	8.3	8.4	7.5
10/76 to 5/77				
S	8.4	8.4	-	7.8
B	8.2	8.3	-	7.8
6/77 to 8/77				
S	8.5	8.8	-	8.1
B	8.4	8.9	-	7.8

S = Surface (1 m)

B = Bottom (0.5 m from sediments)

Dash (-) indicates not sampled.

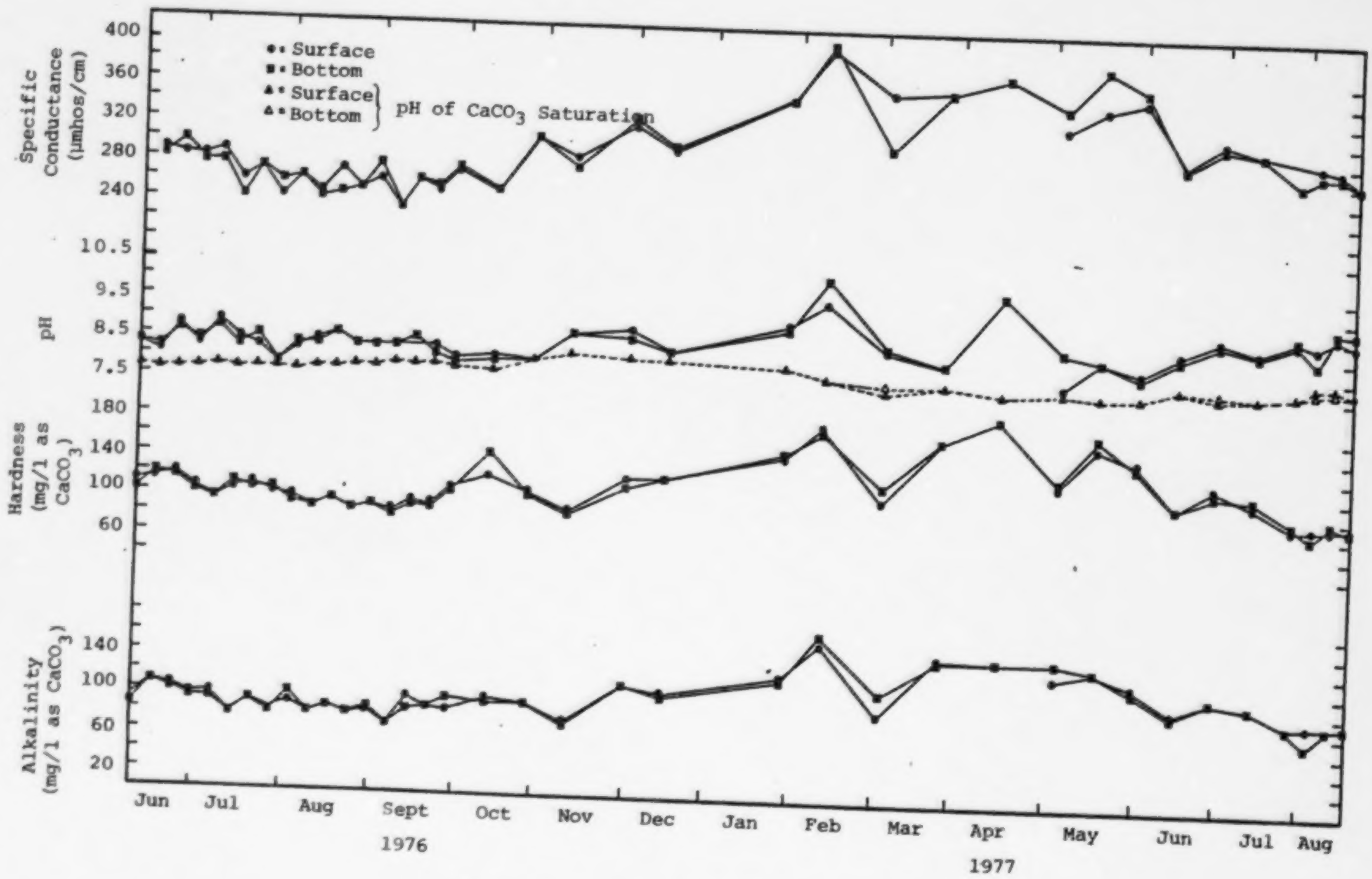


Figure 38. Chemical parameters - Station 9

alkalinity, and total hardness values. The pH was somewhat lower during the spring than in the winter; values ranged from 7.7 to 9 during the summer of 1977 with the pH of the surface and bottom waters being equal. There was an increase in the pH over the summer which corresponded to an increase in the algal numbers and, therefore, to an increase in the uptake of CO_2 by the algae.

Specific Conductance

The specific conductance was lowest during the summer of 1976, increased during the winter months, and then decreased slightly during the summer of 1977. The specific conductance values of the surface and bottom waters were generally about the same due to the wind-induced mixing of the water in this arm of the lake. During the summer of 1976, the specific conductance ranged from 235 to 300 µmhos/cm. During the winter, the specific conductance increased to 410 µmhos/cm, most likely due to the dissolution of $CaCO_3$ from the sediments. There was a decrease during the summer months to a range of 280 to 320 µmhos/cm.

Hardness

The total hardness values in the surface and bottom waters at Station 9 were equal throughout the study as shown in Figure 38. During the summer of 1976, the hardness ranged from 80 to 120 mg/l as $CaCO_3$. During the winter months, the hardness increased and ranged from 90 to 195 mg/l as $CaCO_3$. Since waters are generally considered to be hard if the total hardness is greater than 150 mg/l as $CaCO_3$, Lake Ray Hubbard water can be classified as moderately hard. The total hardness decreased during the summer of 1977 and ranged from 80 to 130 mg/l as $CaCO_3$. The lower hardness during the summer months indicates the possibility of $CaCO_3$ precipitation. Soluble ortho P could be removed from the water column by coprecipitation with $CaCO_3$ which would reduce the amount of P available for algal growth during the summer months.

Alkalinity

The alklainity values in the surface and bottom waters at Station 9 are shown in Figure 38. As with the other chemical parameters measured at Station 9, the alkalinity values of the surface and bottom waters were essentially equal throughout the study. During the summer of 1976, the alkalinity ranged from 70 to 100 mg/l as $CaCO_3$. During the winter months, the alkalinity increased, most likely due to $CaCO_3$ dissolution from the sediments, and ranged from 75 to 170 mg/l as $CaCO_3$. There was a decrease in the alkalinity over the summer of 1977 from 125 mg/l as $CaCO_3$ at the end of May to 75 mg/l as $CaCO_3$ in August. The lower alkalinity values during the summer were likely due to $CaCO_3$ precipitation.

Chloride

The average chloride concentrations in Rowlett Arm are shown in Table 16. The decrease in the chloride concentrations between Station 14 where Rowlett Creek enters the lake, and Station 8, near the Main Body of the lake was primarily due to mixing of the Rowlett Arm waters with the Main Body lake water. These results showed that one volume of water at Station 14 would have had to be diluted with 9 to 18 volumes of Station 7 water in order to obtain the chloride concentrations found at Station 8. These numbers were obtained by calculating the volume of water with the Station 7 chloride concentration (during each period of thermal stratification and during the isothermal period) that would be required to dilute one volume of water with the Station 14 chloride concentration to one volume of water with the Station 8 chloride concentration. During the summer of 1976, the chloride concentration ranged from 17.8 mg/l at Station 8 to 46.9 mg/l at Station 14. During the winter months, the concentrations were lower and the range (13.5 to 26.8 mg/l) was smaller. The concentrations increased by the summer of 1977 and ranged from 16.6 to 38.6 mg/l. The higher concentrations found during the summer months were due to the fact that the flows during

TABLE 16. AVERAGE CHLORIDE - ROWLETT ARM

Depth	Station 8	9	10	14
6/76 to 9/76				
S	17.8	19.8	20.0	45.3
B	18.6	19.3	19.5	46.9
10/76 to 5/77				
S	13.5	15.4	-	26.8
B	13.8	15.6	-	26.4
6/77 to 8/77				
S	16.7	20.5	-	36.7
B	16.6	21.0	-	38.6

S = Surface (1 m)

B = Bottom (0.5 m from sediments)

Dash (-) indicates not sampled.

the summer were primarily derived from the high chloride concentration sewage treatment plant discharge. For example, the chloride concentration in the City of Garland's domestic wastewater typically ranged from 39 to 76 mg/l (See Appendix).

Secchi Depth

The average Secchi depths and average surface turbidities in Rowlett Arm during the study period are shown in Figure 39. Rowlett Arm had the most turbid water in the lake with the Secchi depth ranging from 0.2 to 0.6 m and the turbidity ranging from 6 to 70 FTU's. The Secchi depth was, in general, inversely related to the turbidity. In Rowlett Arm, the fluctuations in the turbidity were much greater than in the Main Body. The Secchi depth fluctuated from week to week with no apparent trend over the annual cycle. This differs from the Main Body of the lake in which the higher Secchi depths were found during the summer months, and the lower Secchi depths were found during the winter months.

The low Secchi depths found in this part of the lake were due to particulate matter derived from the algae that grow in the water, the detritus formed from the death of algae and other organisms, the allocthonous (external to the lake) detritus, any detritus resuspended from the sediments, and the inorganic matter derived from the sediments and tributaries.

Light Penetration

The most turbid water in the lake was found in Rowlett Arm. The light penetration at 1 m ranged from 0.08 percent to 3 percent with an average value of 1 percent. This arm is only a few meters deep so the constant mixing by the wind keeps some of the sediment particles suspended in the water column.

At Station 9, the compensation depth was variable and ranged from 1 to 2 m during the study period. The total primary production on a per unit area basis was lowest in Rowlett Arm. As discussed previously, the nutrient concentrations were often

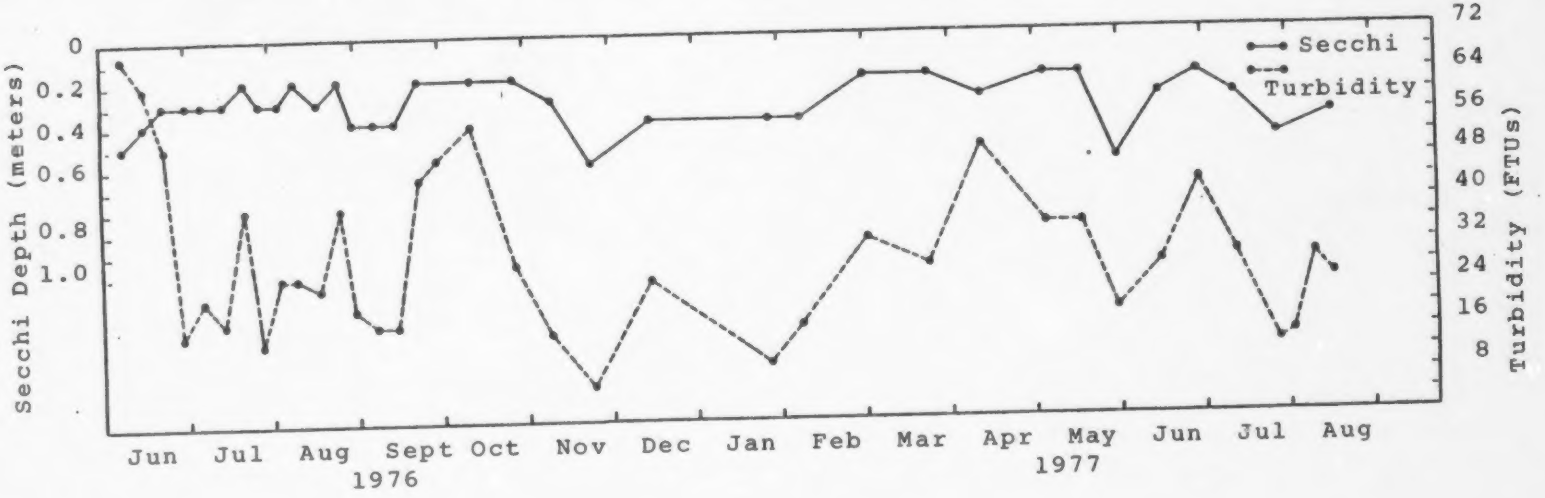


Figure 39. Average Secchi depth and turbidity - Rowlett Arm

high at Station 9; however, since the euphotic zone was so shallow, light may have been limiting algal productivity in this area of the lake at times. The factor relating the compensation depth to the Secchi depth was found to be 3.7 in Rowlett Arm.

The extinction coefficient varied from 3.4 m^{-1} to 8 m^{-1} at Station 9 in Rowlett Arm as shown in Figure 40 . The variability was likely due to the wind and organism-induced suspension of the sediments. The relationship between the Secchi depth and the extinction coefficient in Rowlett Arm during the first year of the study is given by the following equation:

$$Ke = \frac{2.0}{(\text{Secchi})^{0.8}}$$

The relationship between the uncorrected chlorophyll a concentration and the extinction coefficient is given by the following equation:

$$Ke = 3.3 + 0.02 \times [\text{uncorrected chlorophyll } \underline{a} \ (\mu g/l)]$$

The extinction coefficient due to natural turbidity and factors other than the algal concentration (3.3) was much larger in Rowlett Arm than in the Main Body of Lake Ray Hubbard. This indicates that the Secchi depth was controlled by factors other than the algal population to a greater extent in Rowlett Arm.

Algae

The algal population of Rowlett Arm fluctuated throughout the year as shown in Figure 41. The largest populations were found during July and November of 1976, and January, May, and July of 1977. The peak of 23,000 organisms/ml during the summer of 1976 was due to a large increase in the population of the diatom, Cyclotella. There was not a significant increase in the blue-green algal population over the summer in Rowlett Arm as there was in the Main Body. However, the populations did reach a few thousand organisms/ml during late summer. The large

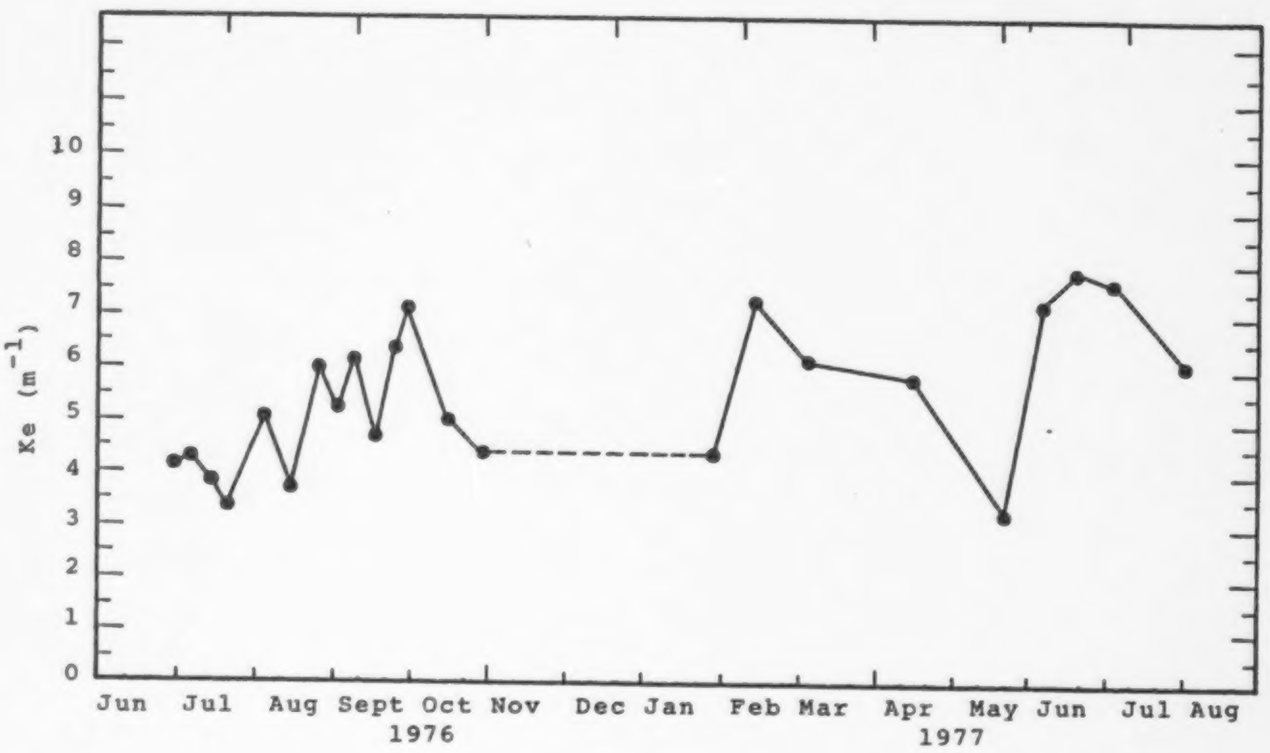


Figure 40. Extinction coefficient - Station 9

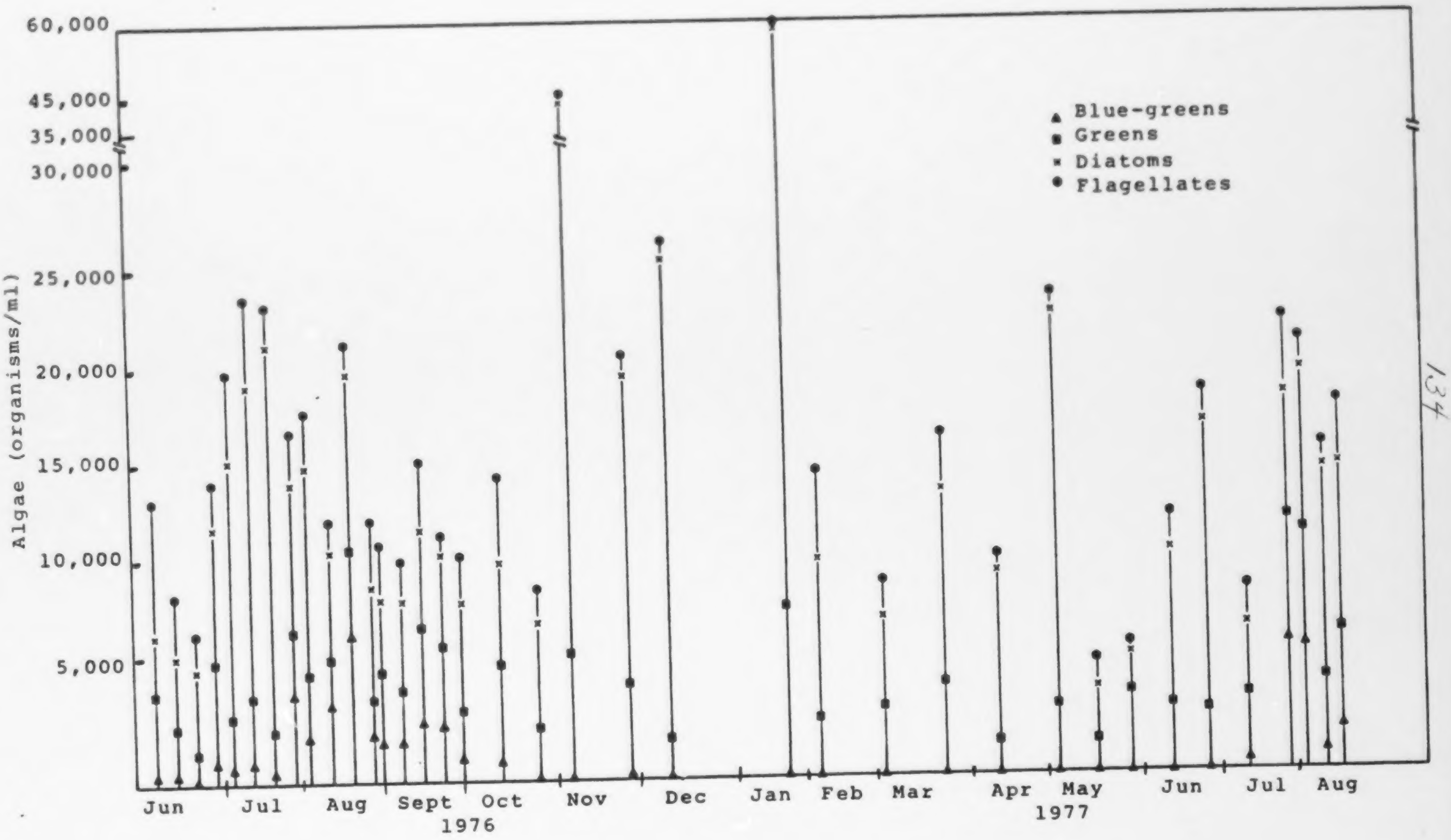


Figure 41. Algal numbers and types - Rowlett Arm

influx of nutrients from Rowlett Creek during the spring was likely responsible for the algal bloom since the soluble ortho P and inorganic N concentrations were both high during June.

There was a large bloom of Cyclotella at Station 9 during November with the total number of algae reaching 77,600 organisms/ml at this station. The average number in Rowlett Arm at this time was 43,000 organisms/ml. The algal population remained large throughout December and January. In January, there was another large bloom of Cyclotella with the average total number of algae reaching 60,000 organisms/ml. The winter blooms were likely due to the large increase in the nitrogen concentration in Rowlett Arm at this time, since nitrogen had been limiting algal growth.

At the beginning of May, there was an increase in the total number of algae to 23,000 organisms/ml due to a bloom of Cyclotella. This could be a response to the large influx of nutrients during the spring and the increasing temperature and photoperiod. The algal population decreased during mid-May and then started to increase over the summer to reach a peak of 22,000 organisms/ml in July.

The algal population was larger in Rowlett Arm than in the Main Body due to the large amounts of nutrients entering in Rowlett Creek. The average total number of algae ranged from 5,500 to 60,000 organisms/ml. Unlike the pattern of algal numbers found in the Main Body, the total number of algae in Rowlett Arm fluctuated from week to week without a definite cycle. Due to the high soluble ortho P and inorganic N concentrations found from December to May 1977 and the amount of suspended material in the water column, light rather than nutrients was most likely controlling algal growth during the winter months. Another difference between Rowlett Arm and the Main Body is the species composition of the population. In Rowlett Arm, the diatoms dominated the population over the annual cycle; whereas in the Main Body, the green and blue-green algae were dominant.

The Rowlett Arm algal population differed from the classical textbook description of algal numbers and types over the annual cycle and with increasing fertility. Normally, the diatoms would be prevalent during the fall, winter, and spring months, followed by green and blue-green algal dominance through the late spring and summer. It might be expected that as the overall fertility of a water body, or part thereof, increased, there would be a shift from diatoms to blue-greens. If Lake Ray Hubbard behaved normally with respect to the algal numbers and types during the two summers that were investigated, then further work needs to be done to ascertain why the types of algae in this lake during this period do not follow the classical patterns found for many other lakes.

Chlorophyll a

Rowlett Arm had the highest concentrations of uncorrected and corrected chlorophyll a in the lake throughout the study. The average uncorrected chlorophyll a concentration fluctuated from week to week and ranged from 13 to 118 µg/l as shown in Figure 41. Although the concentrations were typically high, and there were large fluctuations, the two highest concentrations (118 and 110 µg/l) were found during January and August, respectively, indicating that there were large blooms of algae in which the biomass was significantly larger than normal at these times. As shown in Figure 42, there was an algal bloom in January and the algal numbers were increasing in August. The summer and winter blooms were also found in the Main Body and other areas of Lake Ray Hubbard, although the concentrations of chlorophyll a and the number of algae in Rowlett Arm were much greater than those in the other areas of the lake.

Normally about 50 to 80 percent of the uncorrected chlorophyll a was due to chlorophyll a with pheophytin a making up the remaining 30 to 50 percent of the total. The chlorophyll a

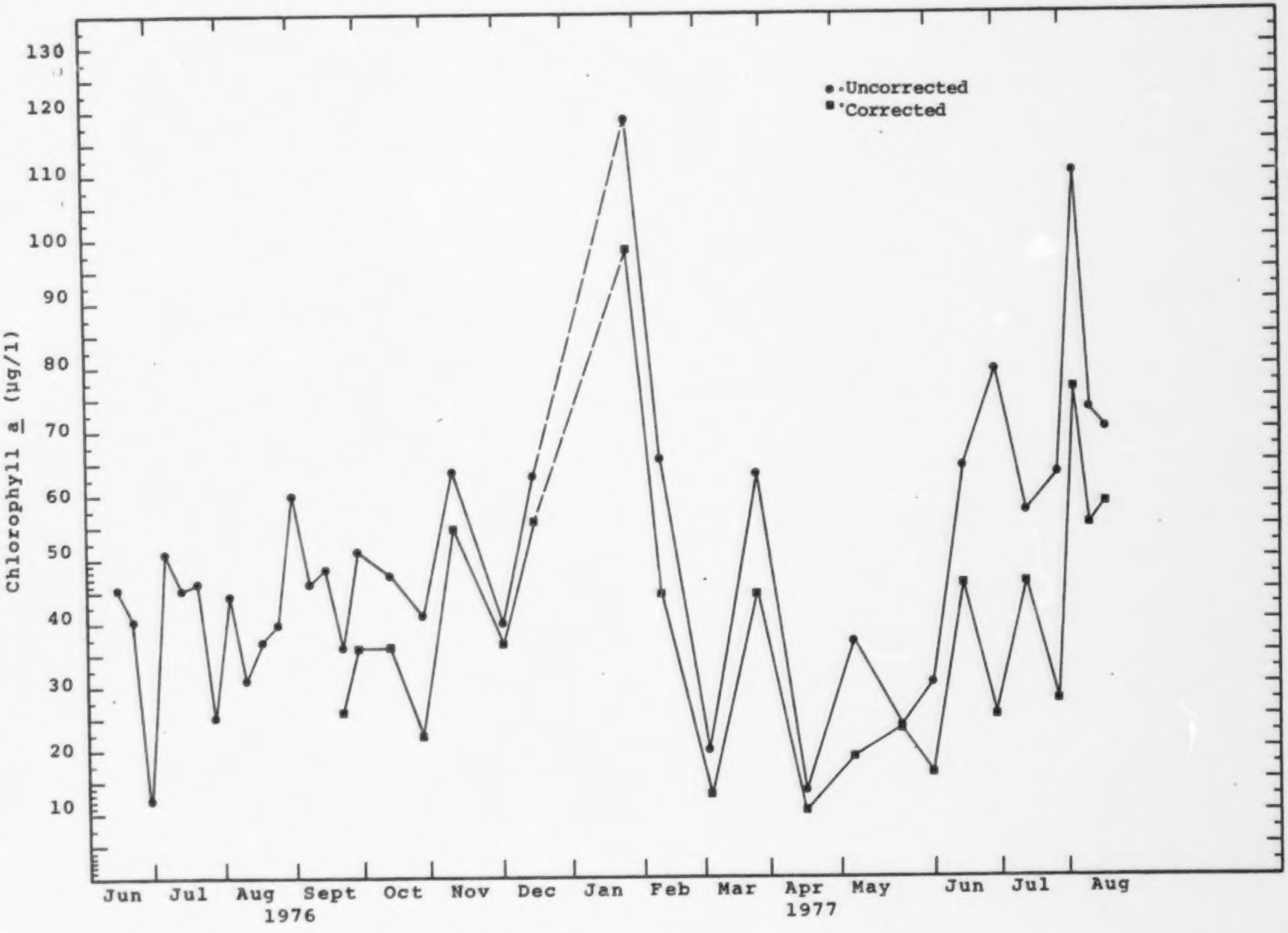


Figure 42. Chlorophyll a concentration - Rowlett Arm

concentrations showed the same fluctuations as the uncorrected chlorophyll _a_, with peaks in August and January.

Primary Productivity

Rowlett Arm was the area with the lowest total production per square meter per hour in Lake Ray Hubbard. The primary productivity ranged from 177 to 367 mg C/m^2/hr in Rowlett Arm which was lower than the productivities of 266 to 596 mg C/m^2/hr in the Main Body. The highest productivities were found during the summer months and the lowest, during the fall and spring months in both areas of the lake. Rowlett Arm had the highest algal concentrations in the lake throughout the study, but primary production was limited by light since the euphotic zone was only 1.5 to 2 m deep. Therefore, even though Rowlett Arm had a much higher algal biomass, the biomass was relatively inactive compared to that in other parts of the lake.

Threshold Odor Number

The average threshold odor numbers in Rowlett Arm are shown in Figure 43. The T.O.N. ranged from 7 to 18 with large fluctuations from week to week. There was no apparent relationship between the T.O.N. and the number of algae in Rowlett Arm. This may be due to the low numbers of blue-green algae which release substances to the water which are odorous in themselves, can be made odorous during water purification, or which stimulate growth of actinomycetes. The odor characteristics most commonly found were musty, earthy, grassy, and rooty with an occasional septic odor.

Actinomycetes

The numbers of actinomycetes at Station 9 in Rowlett Arm are shown in Figure 44. The numbers were much higher than those in the Main Body, and ranged from 7 to 80 organisms/ml. As with the Main Body, there was no apparent relationship between the algal numbers or T.O.N. and the number of actinomycetes. The blue-green algae, which are thought to produce compounds which stimulate the growth of actinomycetes, were found in low numbers in

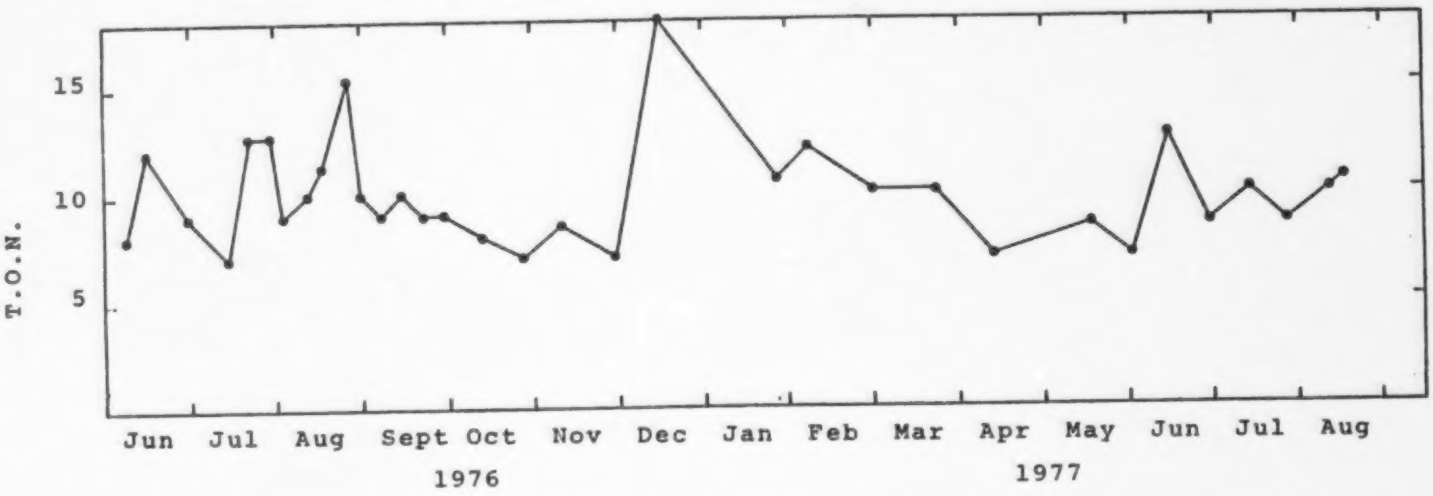


Figure 43. Threshold odor number - Rowlett Arm

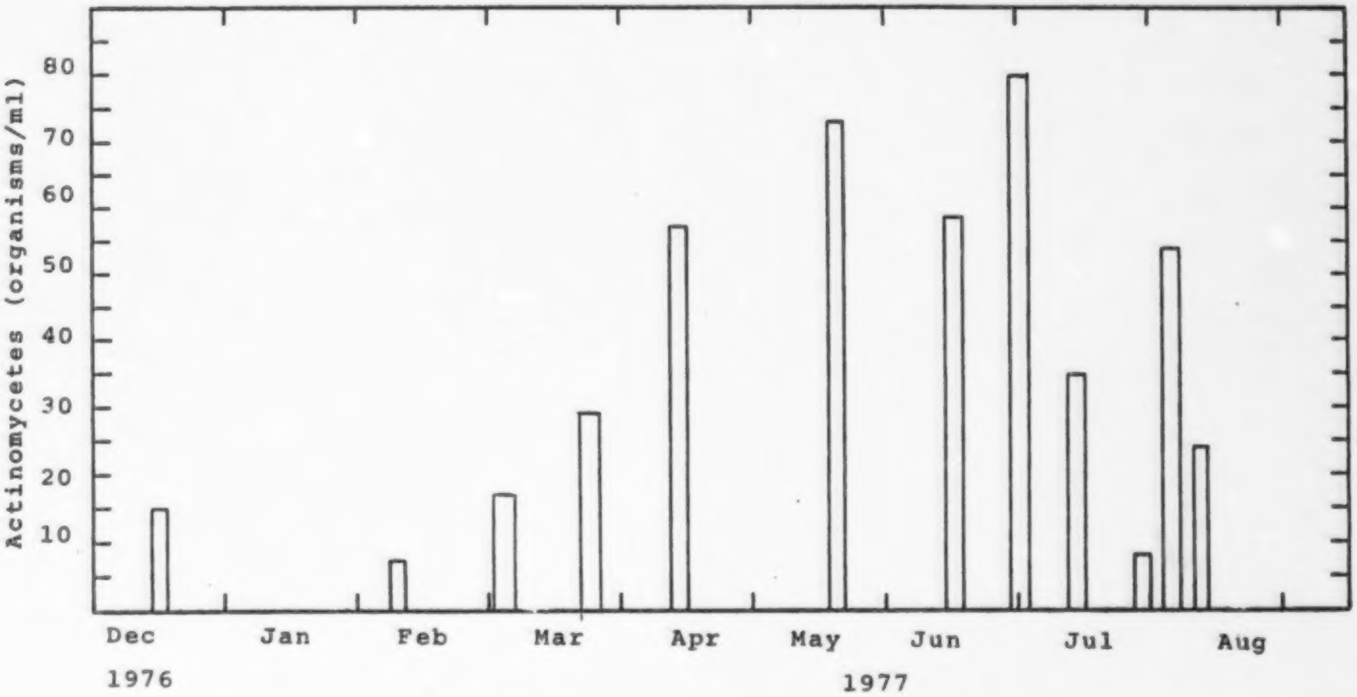


Figure 44. Actinomycetes - Station 9

Rowlett Arm, while the number of actinomycetes was large. It appears that the statement in Standard Methods (APHA *et al.*, 1976) concerning factors controlling the growth of actinomycetes, i.e., the excretion of organics by blue-green algae, should be expanded to include other algae and possibly allocthonous sources of organics.

Currents

Current measurements were made at Stations 8 and 9 in Rowlett Arm on every sampling trip from March 22, 1977 to August 16, 1977. Due to the shallowness of this area of the lake, currents were only measured at a depth of 0.5 m. Table 17 contains the current data for Rowlett Arm. The currents were normally flowing to the north, northeast, or northwest, as in the Main Body. On a few days, the currents were flowing to the southwest or west. The velocities ranged from less than the detection limit to 7.1 cm/sec. In general, the Rowlett Arm current velocities were lower than the Main Body velocities; however, there were several days when the Rowlett Arm current velocities were considerably larger than the Main Body velocities.

The Rowlett Arm current measurements show that in spite of the flow of Rowlett Creek, the water in Rowlett Arm was normally not moving toward the Main Body of the lake at the time of current measurements. This further supports the significance of wind as a factor in controlling currents in Lake Ray Hubbard. The wind and organism-induced suspension of the sediments can have several effects on nutrient load-response relationships in Lake Ray Hubbard. These are discussed in a subsequent section of this report.

Overall Characteristics of Rowlett Arm

From an overall point of view, Rowlett Arm is a much more fertile (hypereutrophic) body of water than the Main Body of the lake and has significantly degraded water quality which impairs its recreational uses. This is to be expected since approximately 53 percent of the N and 74 percent of the P enter the lake via

TABLE 17. LAKE RAY HUBBARD CURRENT MEASUREMENTS ROWLETT ARM

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
March 22, 1977				
8	11:50	0.5	1.7	*
		1.5	0.4	*
9	12:20	0.5	3.5	35
April 12, 1977				
8	12:25	0.5	6	10
9	13:00	0.5	6	10
May 17, 1977				
8	13:05	0.5	4.0	310
9	13:40	0.5	2.9	330
May 31, 1977				
8		0.5	0.8	240
9		0.5	1.5	40
June 14, 1977				
8	12:45	0.5	1.8	210
9	13:12	0.5	-	-
June 28, 1977				
8	14:10	0.5	4.2	330
9	14:35	0.5	7.1	340
July 12, 1977				
8	12:30	0.5	4.6	30
9	13:00	0.5	3.4	0

(Continued)

Table 17 (Concluded)

Station Number	Time	Depth (m)	Velocity (cm/sec)	Direction (degrees from magnetic north)
July 26, 1977				
8	12:35	0.5	3.3	260
9	13:05	0.5	2.0	260
August 2, 1977				
8	11:50	1	-	-
9	12:30	1	-	-
August 9, 1977				
8	12:00	1	3.9	25
9	12:20	1	3.6	335
August 16, 1977				
8	13:10	1	5.2	10
9	13:30	1	3.2	*

Dash (-) indicates no measurable current.

*Indicates not measured.

this arm. Large populations of various types of algae are present in this part of the lake at different times of the year. In general, the blue-green algal numbers are somewhat lower than would be expected based on the overall trophic state of Rowlett Arm. The differences in the types of algae found in this part of the lake compared to the other parts may be related to the fact that essentially all of the flow entering Rowlett Arm during low flow periods, is derived from the Plano and Garland domestic wastewater treatment plants' effluents.

CHARACTERISTICS OF THE EAST FORK ARM

Dissolved Oxygen and Temperature

Figure 45 shows several DO and temperature profiles at Station 4 in the East Fork Arm of Lake Ray Hubbard. The water column was isothermal throughout the study period due to wind-induced mixing of the shallow water. There was no evidence of a thermocline developing in the East Fork Arm during the summer months. As shown in Figure 8, the surface water temperatures in the East Fork Arm, ranging from 4°C in January to 32°C in August, were about the same as those found in the Main Body.

The surface DO ranged from 6 to 20 mg/l and was normally slightly supersaturated or just at saturation due to the algal production of oxygen. During the summer months, the DO concentration was often lower near the bottom due to the high oxygen demand of the sediments; however, it was never completely depleted at the bottom as it was in parts of the Main Body. This is likely due to wind-induced mixing of the water column which prevented the formation of a thermocline.

Phosphorus

The average total P concentrations in the East Fork Arm are shown in Figure 46. The total P concentrations were generally between 0.01 and 0.05 mg-P/l with peaks in August and September 1976 and March and August 1977. The August 1976 peak of 0.14 mg-P/l and the September 1976 peak of 0.1 mg-P/l were coincident with high algal numbers and low concentrations

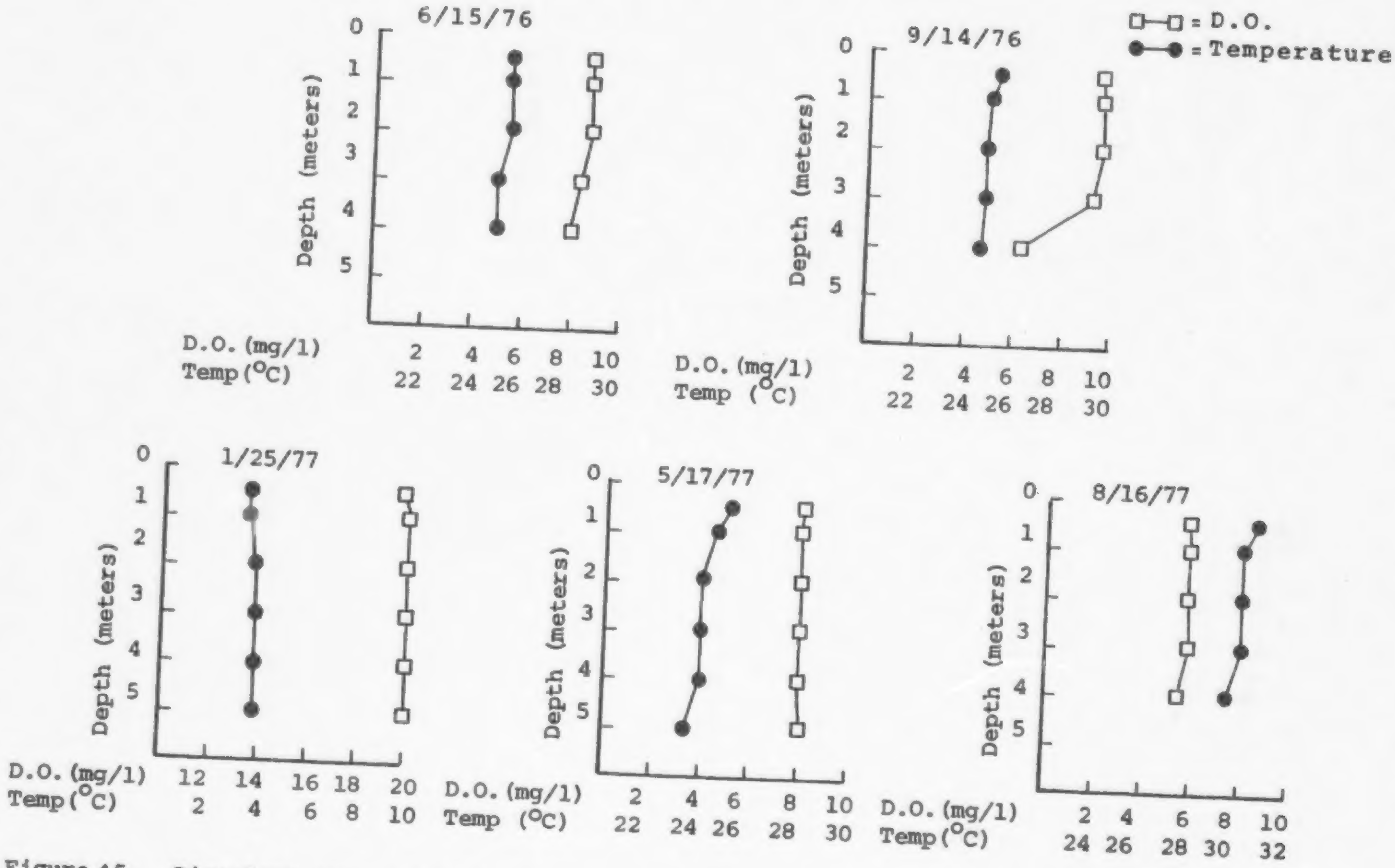


Figure 45. Dissolved oxygen and temperature profiles - Station 4

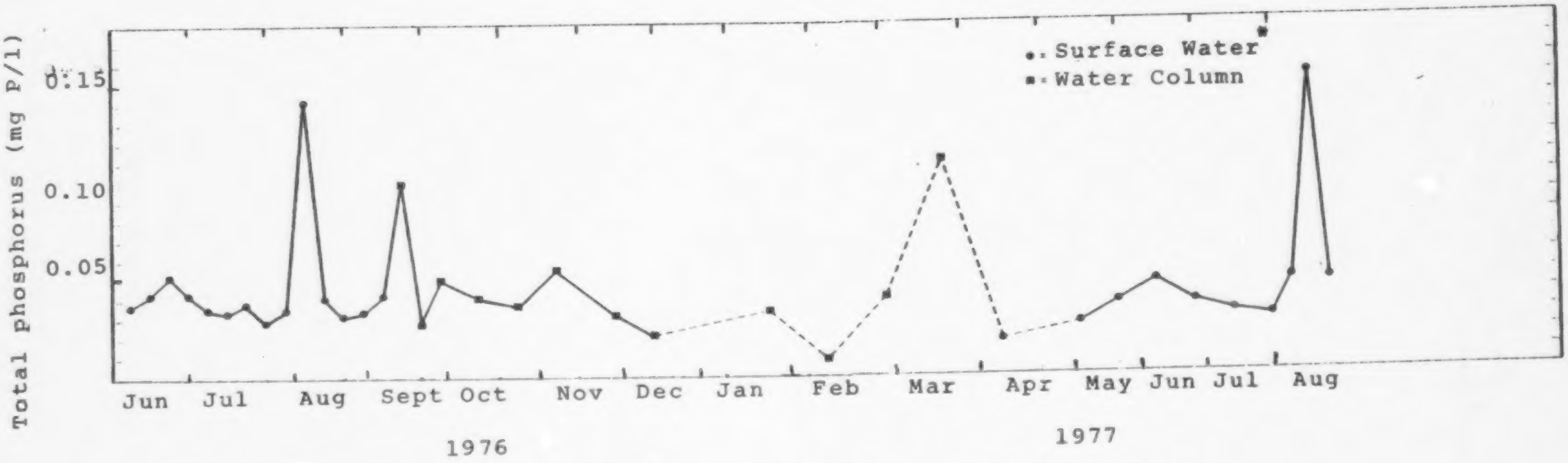


Figure 46. Average total phosphorus concentration - East Fork Arm

*Includes the mid depth sample at Station 3.

of soluble ortho P, indicating that most of the phosphorus was contained in algal cells at these times. The March peak of 0.11 mg-P/l was most likely due to land runoff and recycling from the sediments since the algal population was small and the soluble ortho P concentration was high. In August 1977, the peak of 0.15 mg-P/l was comprised primarily of phosphorus which had been incorporated into the algal cells. In general, the total P concentrations in the East Fork Arm were similar to those found in the Main Body. The total P in both areas of the lake was normally near 0.04 mg-P/l with peaks of about 0.15 mg-P/l.

The average soluble ortho P concentrations are shown in Figure 47. The concentrations in the East Fork Arm ranged from less than the detection limit to 0.009 mg-P/l throughout most of the year with peaks of 0.02 mg-P/l in June 1976, 0.017 mg-P/l in June 1977, and 0.06 mg-P/l in August 1977. In general, the soluble ortho P concentrations were low when the algal numbers were high, and the concentrations were high when the algal population was small, indicating that soluble ortho P was most likely the algal growth limiting nutrient. The soluble ortho P concentrations were generally less than 0.005 mg-P/l in both the Main Body and the East Fork Arm of Lake Ray Hubbard; however, the peak concentrations in the East Fork Arm were higher than the Main Body peaks. This may be due to the input of domestic wastewaters from the City of Rockwall via Squabble Creek.

Nitrogen

The average total N concentrations in the East Fork Arm are shown in Figure 48. There was a high concentration of 4.33 mg-N/l during June of 1976, most likely due to the large inputs of N from the tributaries during the spring of 1976. From mid-July to December, the total N concentration was near 1.0 mg-N/l. During the spring of 1977, there was a large influx of nitrogen from the tributaries which produced a peak of 1.6 mg-N/l during April. After this the concentration decreased to near 0.5 mg-N/l during the summer of 1977. The total N in the

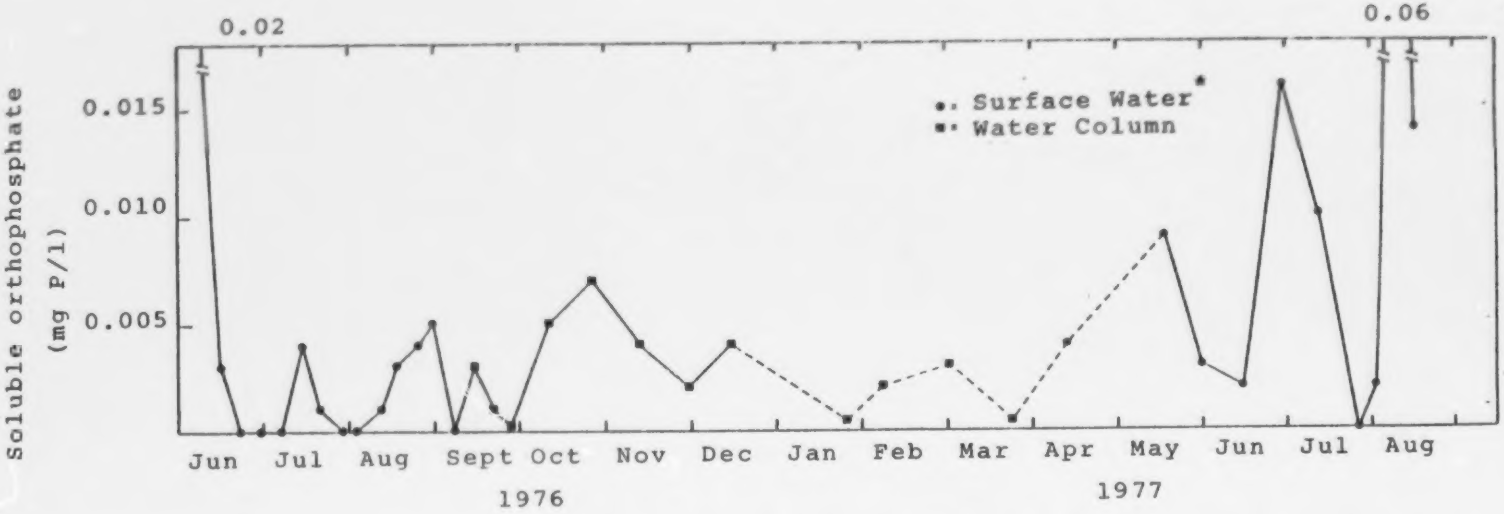


Figure 47. Average soluble orthophosphate concentration - East Fork Arm

*Includes the mid depth sample at Station 3

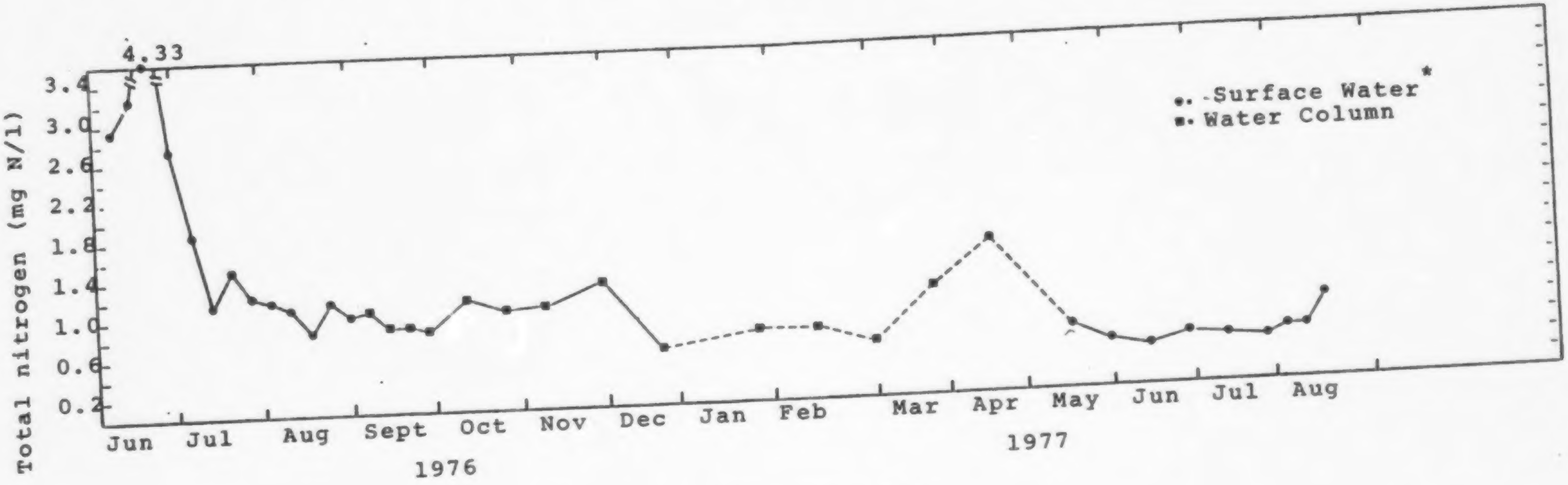


Figure 48. Average total nitrogen - East Form Arm

*Includes the mid depth sample at Station 3

East Fork Arm showed the same trend as in the Main Body of Lake Ray Hubbard. The peak concentrations occurred at the same time and were of about the same magnitude in both areas. Since there was essentially no flow in the East Fork of the Trinity River during the study period, there were no major tributaries contributing nitrogen to the East Fork Arm. However, the smaller tributaries contributed enough N to cause the concentrations in the lake to increase during March and April.

The inorganic N concentrations are shown in Figure 49 . If N were limiting algal growth in this arm of the lake, there would be an inverse relationship between the inorganic N concentration and the total number of algae. During the summer of 1976, the inorganic N concentration did not appear to be related to the number of algae since the inorganic N concentrations were high, reaching peaks of 0.4 and 0.44 mg-N/l when the algal numbers were increasing. However, during the fall and winter months, there appeared to be an inverse relationship between the inorganic N concentration and the algal numbers. During October and November, the concentration increased to 0.20 mg-N/l, most likely due to recycling from the sediments shortly after the lake became isothermal. In the East Fork Arm, there was excess N compared to P at this time with respect to algal requirements. During the winter and early spring months, the inorganic N concentration was near 0.04 mg-N/l. There was an increase in the concentration during April with a peak of 0.25 mg-N/l in mid-May, most likely due to the large tributary input during the spring. The inorganic N concentrations in the East Fork Arm showed the same trends and were of the same magnitude as the inorganic N concentrations in the Main Body.

It appears that the algae in the East Fork Arm were limited by P during the study, so apparently the amount of P in the sewage from Rockwall was not sufficient to drive this part of the lake to N limitation during the study period.

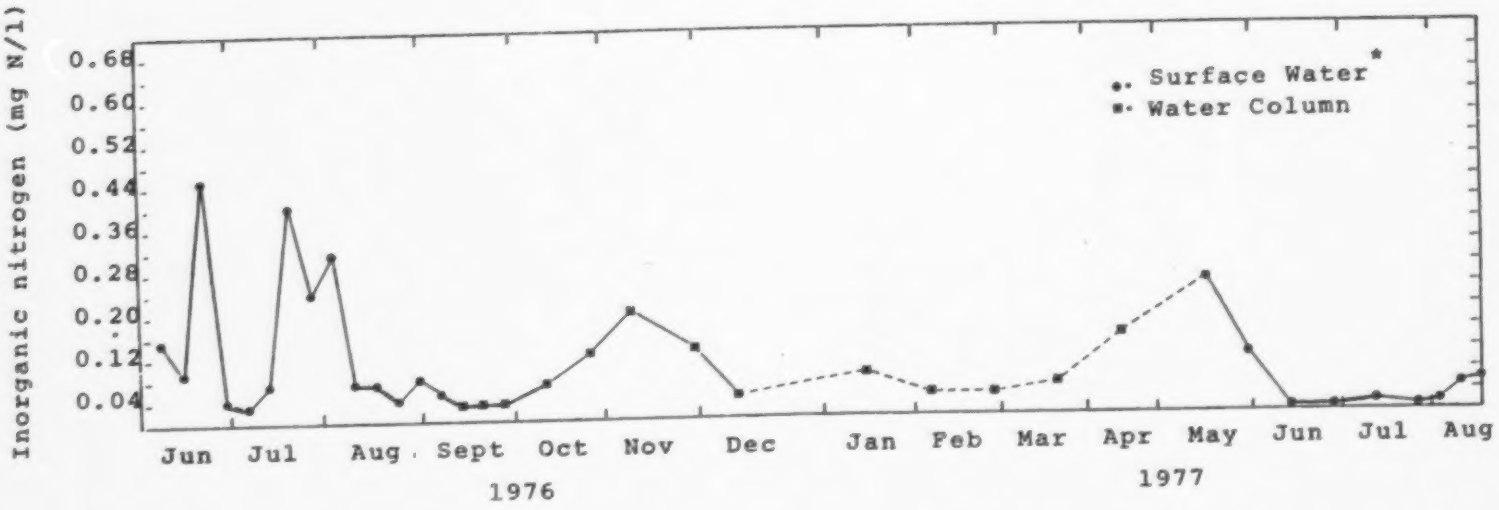


Figure 49. Average inorganic nitrogen concentration - East Form Arm

*Includes the mid depth sample at Station 3

pH

The pH in the East Fork Arm was relatively constant throughout the arm as shown in Table 18. Differences were noted between the summers of 1976 and 1977 and between the summer and winter periods. The pH values were slightly higher than those found in the Main Body, but there was the same increase in the pH values from the summer of 1976 to the summer of 1977.

Figure 50 shows the pH of the surface and bottom waters at Station 4 during the study period. The pH ranged from 7.7 to 8.5 during the summer of 1976 with the highest values found during August when the algal uptake of CO_2 was greatest since the algal populations were largest. During the winter months, the pH ranged from 7.7 to 8.7 but was normally near 8.0. The pH ranged from 8.1 to 8.5 during the summer of 1977. The higher pH during the summer of 1977 could be due to the decreased solubility of CO_2 due to the higher surface water temperatures found in 1977 compared to the summer of 1976.

Specific Conductance

As shown in Figure 50, the specific conductances of the surface and bottom waters at Station 4 were essentially equal throughout the study. During the summer months, the specific conductance was lower than during the winter months, most likely due to $CaCO_3$ precipitation. The specific conductance ranged from 245 to 280 µmhos/cm during the summer of 1976 and from 240 to 275 µmhos/cm during the summer of 1977. During the winter months, the specific conductance was typically near 280 µmhos/cm and ranged from 275 to 300 µmhos/cm. These values are comparable to the surface water values in the Main Body but are somewhat lower than the bottom water values during the summer months.

Hardness

The total hardness in the surface and bottom waters at Station 4 is shown in Figure 50. The surface and bottom water values were essentially equal throughout the study period. This

TABLE 18. AVERAGE pH - EAST FORK ARM

	Station		
Depth	3	4	5
6/76 to 9/76			
S	8.0	8.1	8.1
M	8.0	-	-
B	7.9	8.0	7.9
10/76 to 5/77			
S	8.0	8.1	-
B	8.2	8.2	-
6/77 to 8/77			
S	8.3	8.4	-
M	8.3	-	-
B	8.2	8.4	-

S = Surface (1 m)
M = Mid
B = Bottom (0.5 m from sediments)

Dash (-) indicates not sampled.

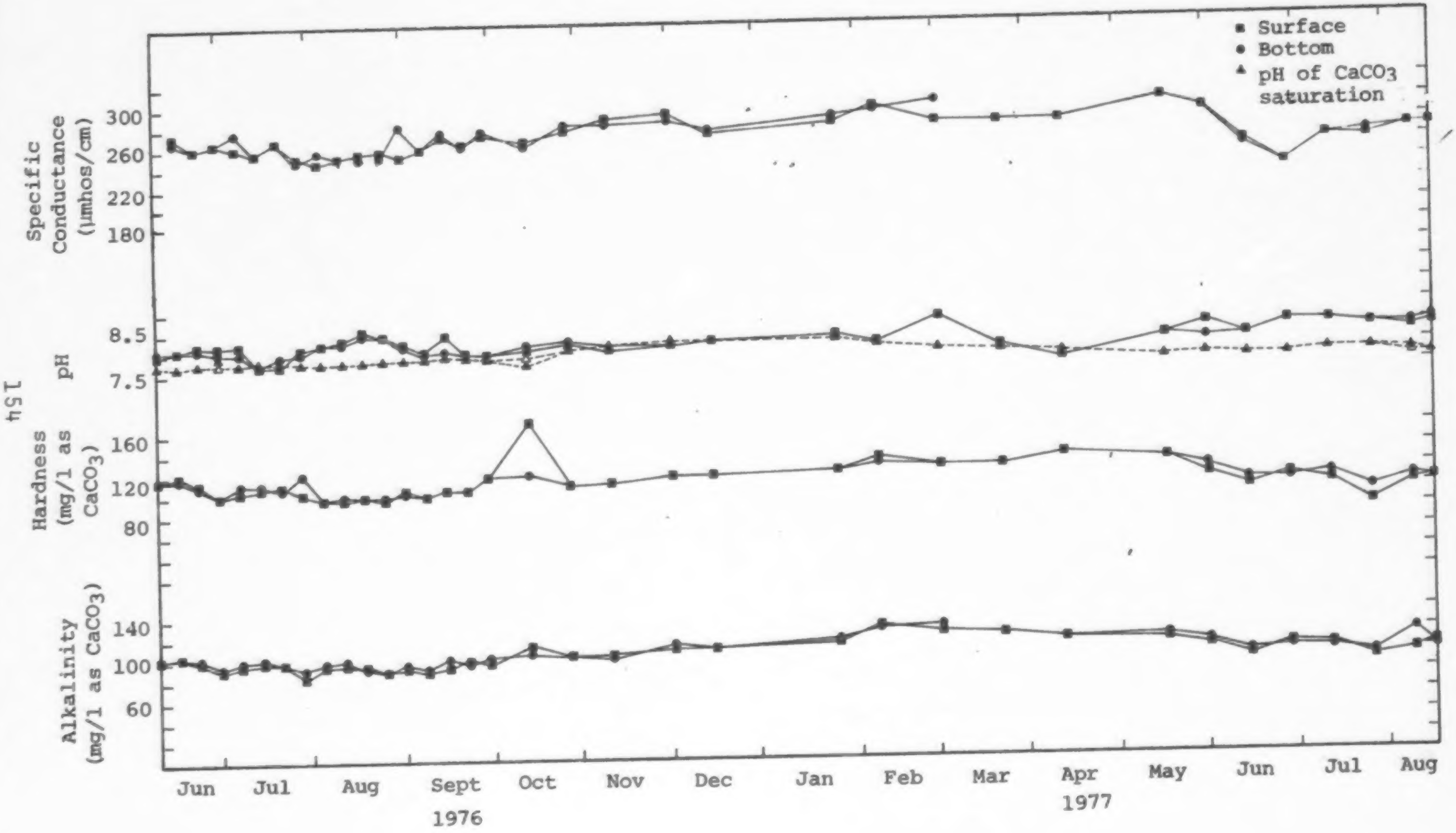


Figure 50. Chemical parameters - Station 4

differed from the Main Body where the hardness of the bottom waters was greater than the surface waters during the periods of thermal stratification. The hardness ranged from 95 to 120 mg/l as $CaCO_3$ during the summer of 1976 which was comparable to the hardness of the surface waters in the Main Body during the summer months. There was an increase during the winter months when the hardness was typically between 110 and 130 mg/l as $CaCO_3$. This range was slightly higher than the hardness of the Main Body waters at that time. During the summer of 1977, the hardness ranged from 80 to 120 mg/l as $CaCO_3$ which was comparable to the Main Body surface water. In the East Fork Arm, as in other parts of Lake Ray Hubbard, the lower summer values compared to the winter values could be due to $CaCO_3$ precipitation.

Alkalinity

The alkalinities of the surface and bottom waters at Station 4 were essentially equal throughout the study period as shown in Figure 50. This differs from the Main Body where the alkalinity of the bottom waters was higher than that of the surface waters during the summer months. As in the other areas of the lake, the alkalinity was lowest during the summer months, possibly due to $CaCO_3$ precipitation. During the summer of 1976, the alkalinity ranged from 85 to 105 mg/l as $CaCO_3$. There was an increase during the winter months to 100 to 130 mg/l as $CaCO_3$. During the summer of 1977, the alkalinity ranged from 95 to 120 mg/l as $CaCO_3$ and was typically near 100 mg/l as $CaCO_3$. The alkalinity in the East Fork Arm was normally slightly higher than the surface water alkalinity but lower than the bottom water alkalinity in the Main Body.

Secchi Depth

The average Secchi depth and values of turbidity in surface water samples from the East Fork Arm are shown in Figure 51. As in the Main Body, the Secchi depth and turbidity were, in general, inversely related. The Secchi depth ranged from

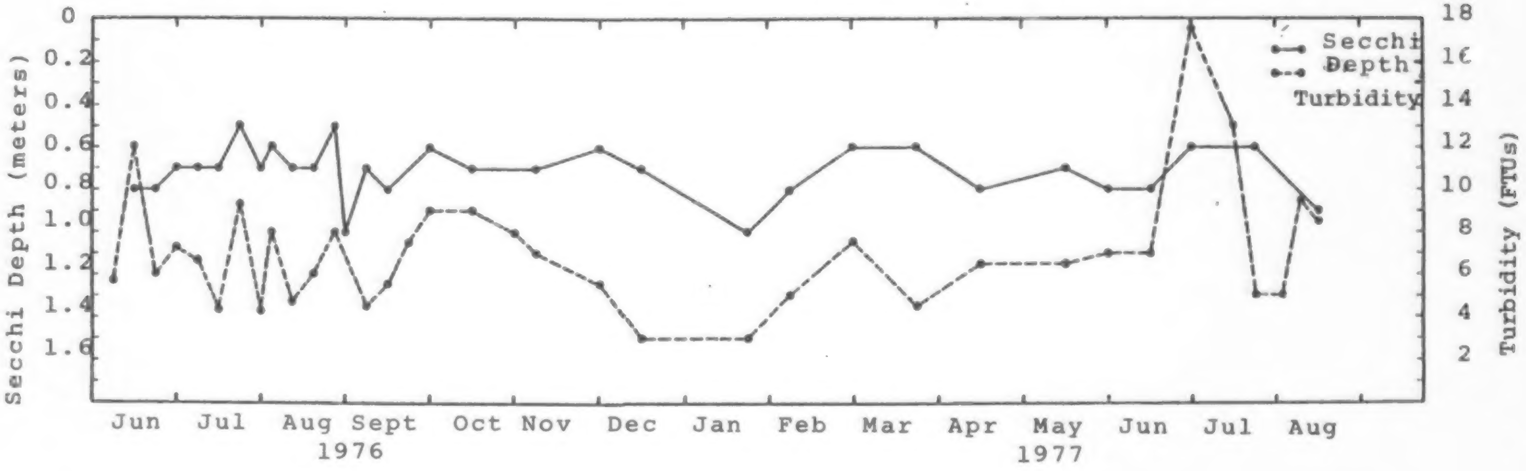


Figure 51. Average Secchi depth and turbidity - East Fork Arm

0.5 to 1 m, and the turbidity ranged from 3 to 18 FTU's. Unlike the Main Body, there were no seasonal trends in the Secchi depth in the East Fork Arm, but there were fluctuations from week to week. Due to the shallowness of this part of the lake, the Secchi depth and turbidity could be influenced by wind-induced suspension of the sediments to a greater degree than in the Main Body. The cause of the very high July 1977 peak in turbidity is unknown.

Light Penetration

The light penetration at 1 m at Station 4 ranged from 4 percent during July 1976 to 30 percent during January 1977. The average value for the study period was 10 percent. The decreased light penetration in this arm compared to that in the Main Body may be due to the higher algal populations and to the shallower depth which results in greater wind mixing of the sediments.

The compensation depth at Station 4 was found to be between 2 and 4 m with an average of 3 m. The relationship between the compensation depth and the Secchi depth in the East Fork Arm is given in the following equation:

$$1\%\ D = 3.8 \times \text{Secchi Depth}.$$

In the East Fork Arm, the extinction coefficient ranged from 0.8 m^{-1} in January to 2.2 m^{-1} during July 1976 and fluctuated from week to week as shown in Figure 52. The average during the study period was 1.5 m^{-1}. The relationship between the Secchi depth and the extinction coefficient in the East Fork Arm was found to be:

$$Ke = \frac{1.3}{(\text{Secchi})^{0.6}}$$

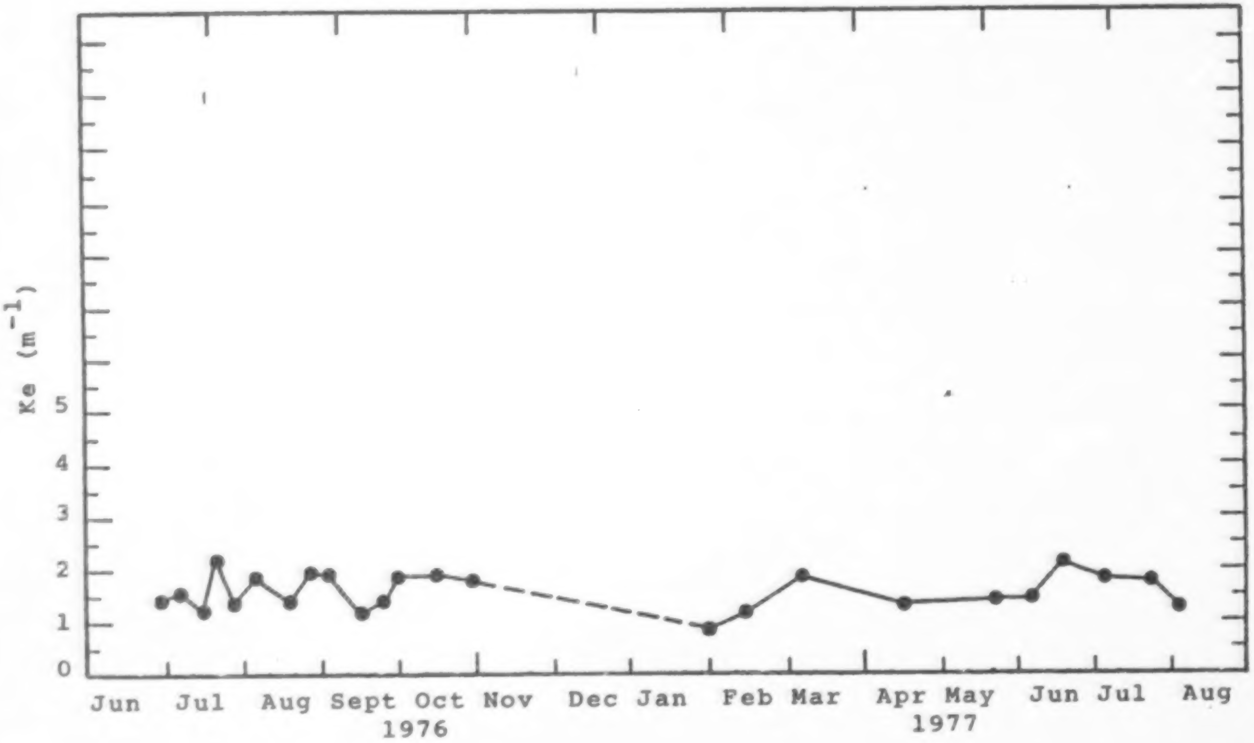


Figure 52. Extinction coefficient - Station 4

The relationship between the extinction coefficient and the uncorrected chlorophyll a concentration is given by the following equation:

$$Ke = 1.5 + 0.0014\ [\text{uncorrected chlorophyll } \underline{a}\ (\mu g/l)]$$

Algae

The algal population in the East Fork Arm showed the same general trend as the Main Body population. As shown in Figure 53, the algal numbers increased during both summers, and there were large peaks in January and February. The number of algae in the East Fork Arm was higher than in the Main Body, but the species composition was similar.

The increase during the summer of 1976 was mainly due to the blue-green alga, *Oscillatoria*. By the end of August, the total number of algae reached 19,000 organisms/ml with 60 percent of the population due to *Oscillatoria*. The nutrient concentrations were high during July so there was sufficient N and P for algal growth during the summer months.

The winter increase started in December and peaked in January and February at 47,000 organisms/ml which was significantly higher than the peak of 22,000 organisms/ml at this time in the Main Body. The increase was due to the diatoms, *Cyclotella* and *Fragillaria*. Since there was not a large input of nutrients from the tributaries at this time, the nutrients for the bloom were most likely recycled from the sediments.

The increase in algal numbers that occurred during the summer of 1977 was due to an increase in *Oscillatoria*. The total number of algae reached 17,000 organisms/ml at the end of July which was comparable to the total number of algae in the Main Body at this time. There was a large input of nitrogen and phosphorus from the tributaries during the spring which supplied the nutrients for the summer bloom.

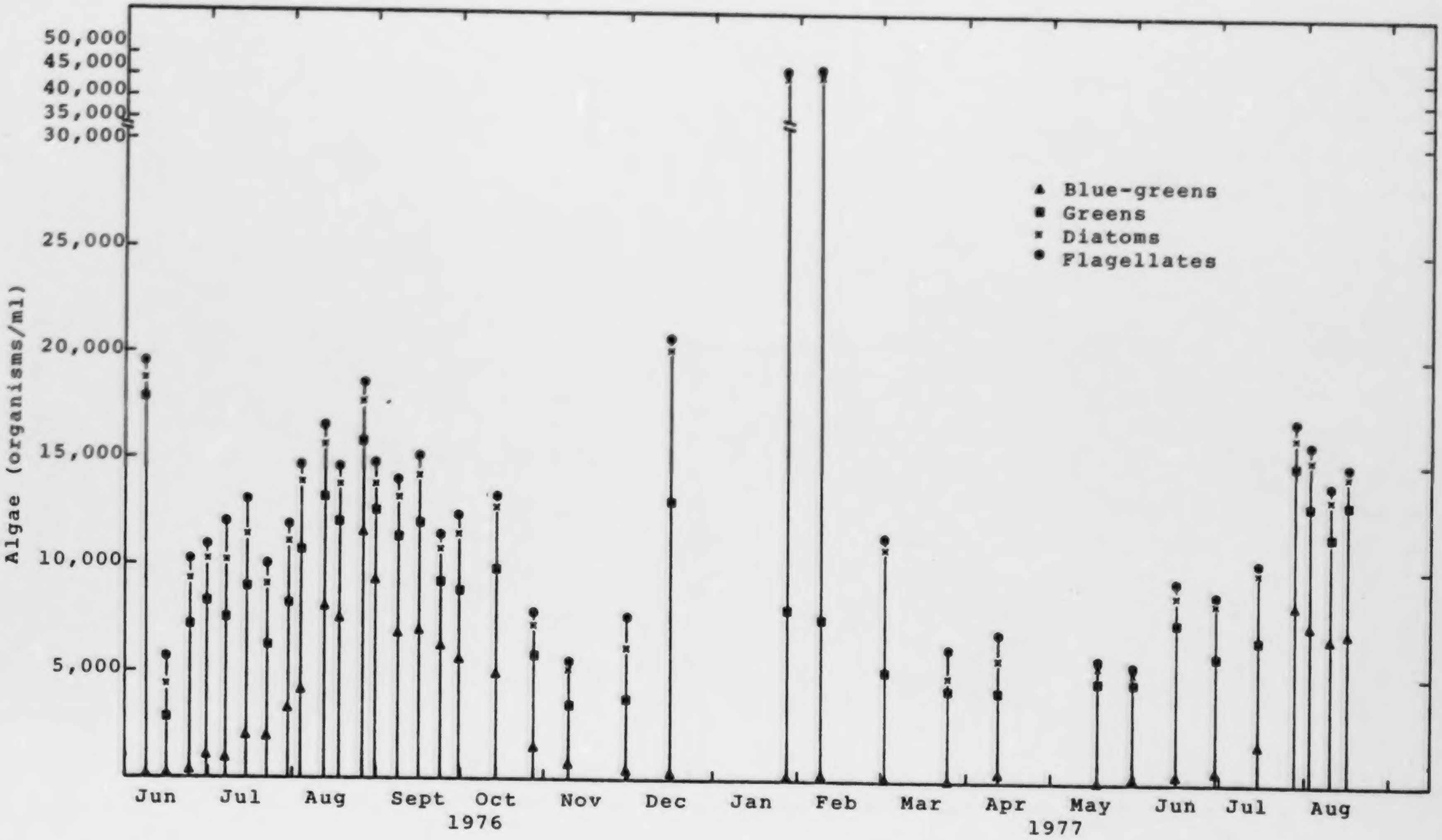


Figure 53. Algal numbers and types - East Fork Arm

Chlorophyll a

The uncorrected chlorophyll a concentrations in the East Fork Arm were generally several µg/l higher than the Main Body concentrations, but the East Fork Arm did not show the same trend as the Main Body which had a summer and winter peak. As shown in Figure 54, the East Fork Arm concentrations fluctuated around 12 µg/l during the summer and fall of 1976 and around 16 µg/l during the summer of 1977. During the winter months, the concentrations were high with a peak concentration of 23 µg/l in March. The uncorrected chlorophyll a concentrations in the East Fork Arm ranged from 6 to 23 µg/l. The chlorophyll a concentrations were generally several µg/l lower than the uncorrected chlorophyll a concentrations and ranged from 4 to 19 µg/l. The pheophytin a concentrations were low compared to the other areas of the lake and ranged from 1 to 5 µg/l.

Primary Productivity

The primary productivity of the East Fork Arm was generally slightly higher on a per unit area basis integrated over the euphotic zone than the productivity of Rowlett Arm and lower than the Main Body productivity. As shown in Table 13, the primary productivity ranged from 238 to 353 mg $C/m^2/hr$ with the highest productivities during the summer and lowest during the fall and spring. The depth of the euphotic zone of the East Fork Arm ranged from 2 to 3 m which was lower than the euphotic zone depth in the Main Body.

Threshold Odor Number

The average threshold odor numbers in the East Fork Arm are shown in Figure 55. During the summer of 1976, the T.O.N. fluctuated between 6 and 15 with musty, grassy, and rooty odor characteristics. The T.O.N. was 7 throughout the winter months with musty and vegetable odor characteristics. During the summer of 1977, the T.O.N. fluctuated from 7 to 10 with musty, vegetable,

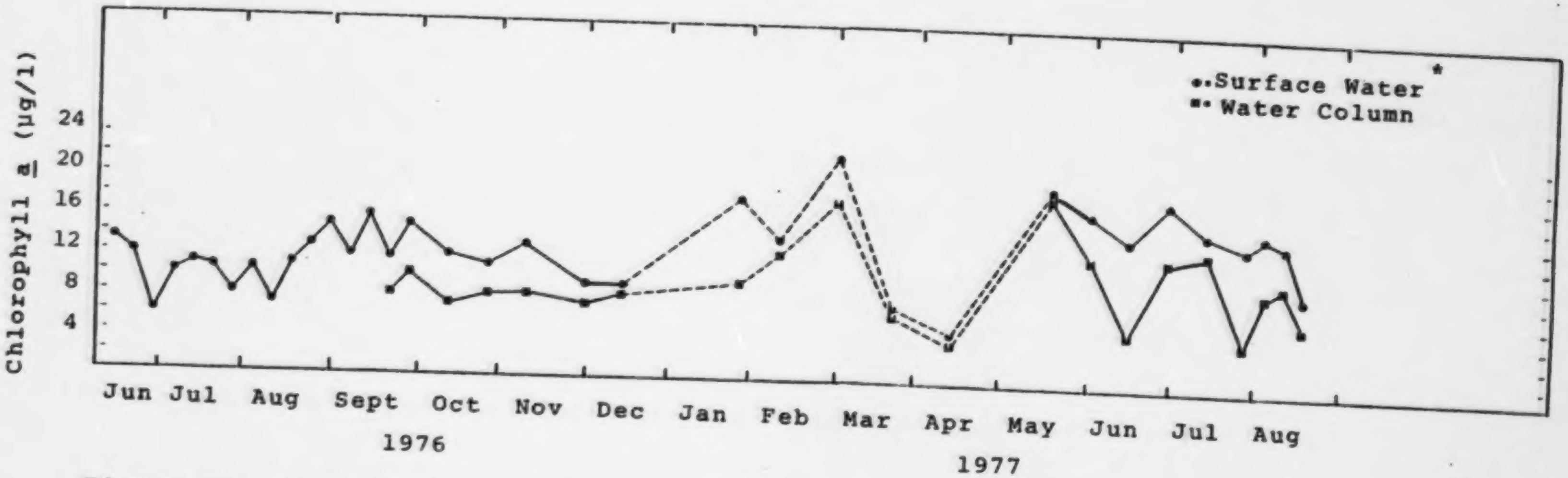


Figure 54. Average chlorophyll a concentration - East Fork Arm

*Includes the mid depth sample at Station 3

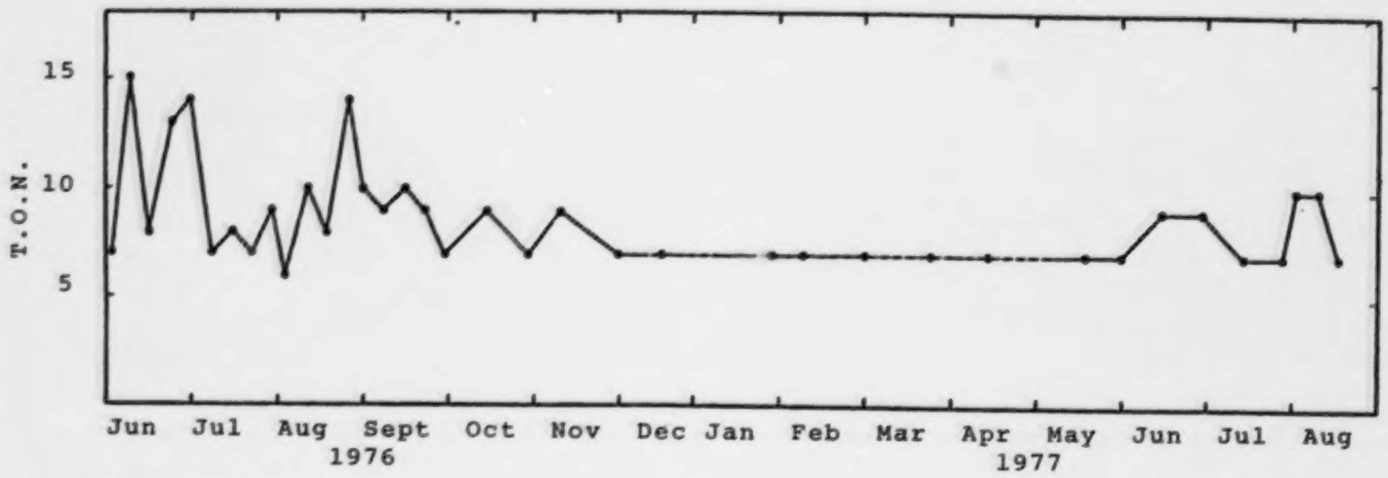


Figure 55. Threshold odor number - East Fork Arm

grassy, and earthy odor characteristics. In general, the T.C.N.'s and odor characteristics in the East Fork Arm were similar to those found in the Main Body.

Actinomycetes

The actinomycetes concentrations at Station 4 during the study period are shown in Figure 56. During December 1976, there were 6 actinomycetes/ml. For the rest of the study period there were only 2 organisms/ml so there was no apparent relationship between actinomycetes numbers and the threshold odor number in the East Fork Arm.

CHARACTERISTICS OF MUDDY CREEK ARM

Dissolved Oxygen and Temperature

The surface water temperatures in Muddy Creek Arm were similar to those of the other areas of the lake, ranging from 6°C in January to 32°C in August. Figure 57 shows several representative DO and temperature profiles at Station 11 in Muddy Creek Arm. In this area of the lake, there was a transient thermocline present during some weeks of the summer in which the temperature changed by 2 to 3°C from surface to bottom. The water was usually completely mixed so the thermocline did not form an effective barrier to mixing of hypolimnetic and epilimnetic waters.

During the winter months, the DO concentration was constant from surface to bottom. Throughout the year, the surface waters were usually saturated or supersaturated with DO due to the photosynthetic production of oxygen by algae. The bottom waters had lower concentrations of DO during the summer months, most likely due to the sediment oxygen demand and the bacterial decomposition of organic matter.

Phosphorus

The average total P concentrations and the average soluble ortho P concentrations in Muddy Creek Arm are shown in Figures 58 and 59 . The total P concentration was generally near 0.08 mg-P/l with peaks during August and October 1976 and

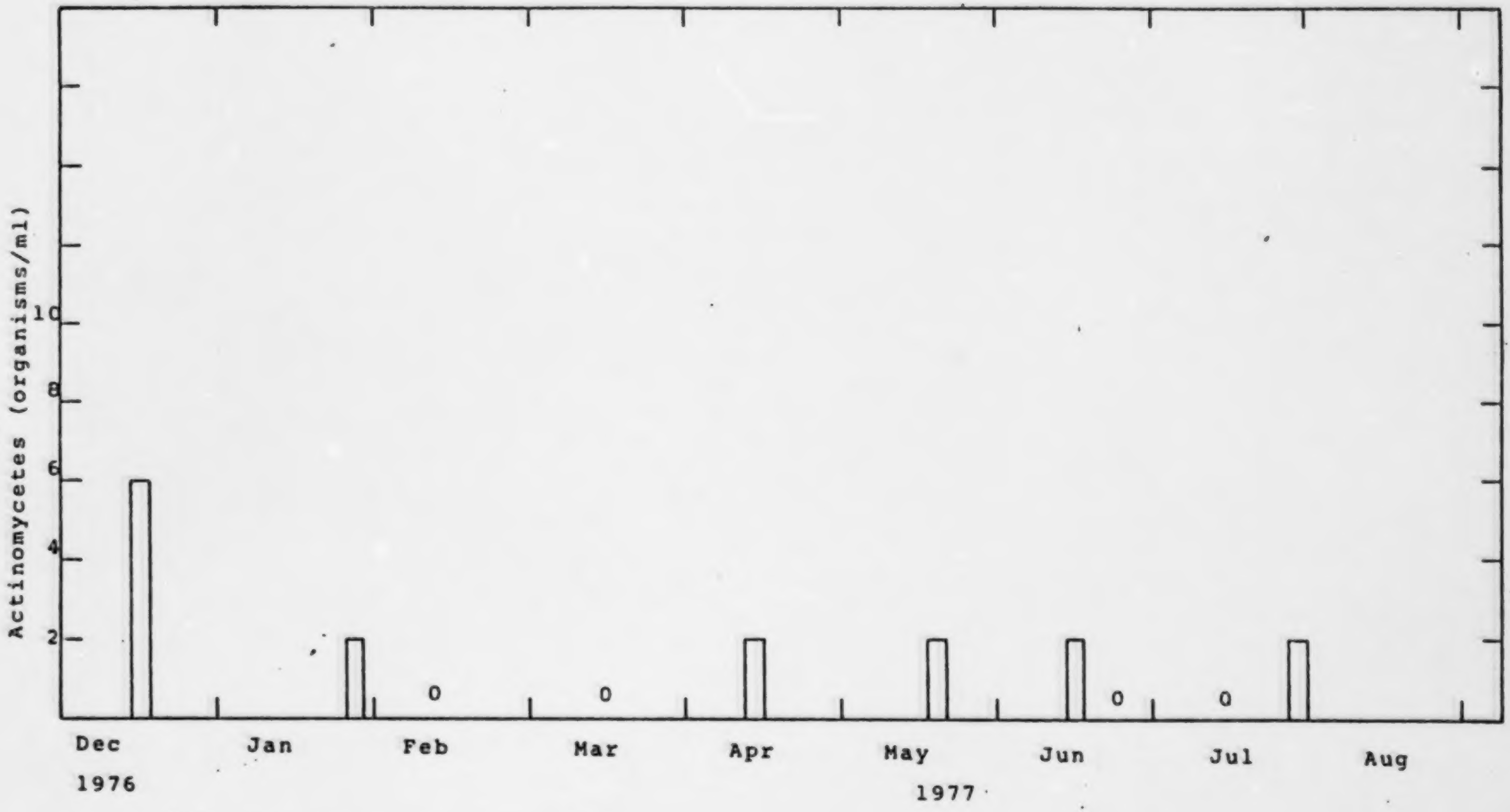


Figure 56. Actinomycetes - Station 4

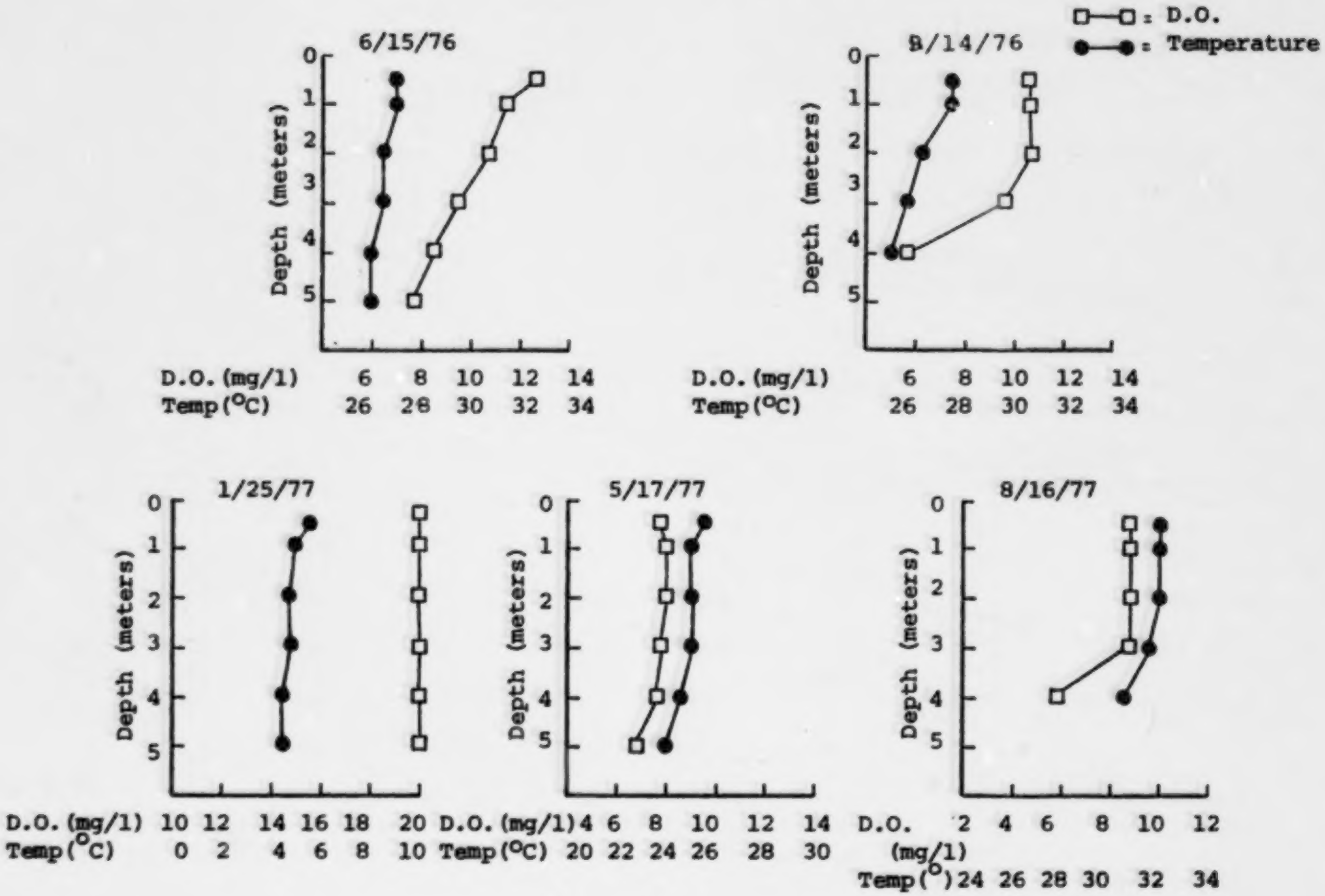


Figure 57. Dissolved oxygen and temperature profiles - Station 11

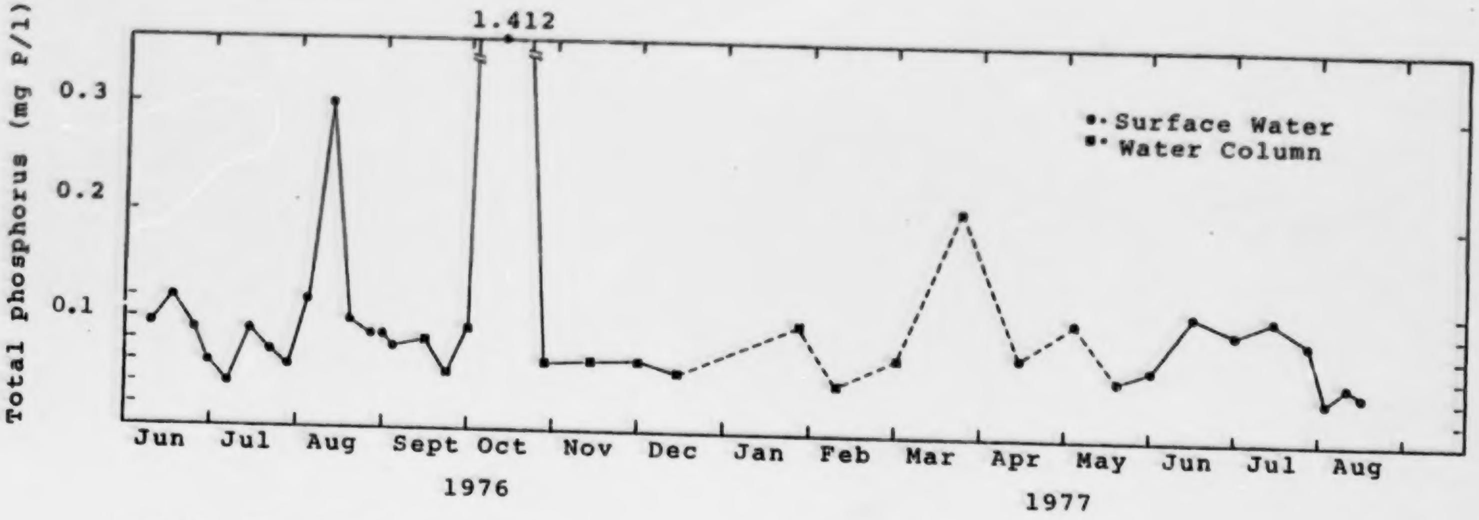


Figure 58. Average total phosphorus concentration - Muddy Creek Arm

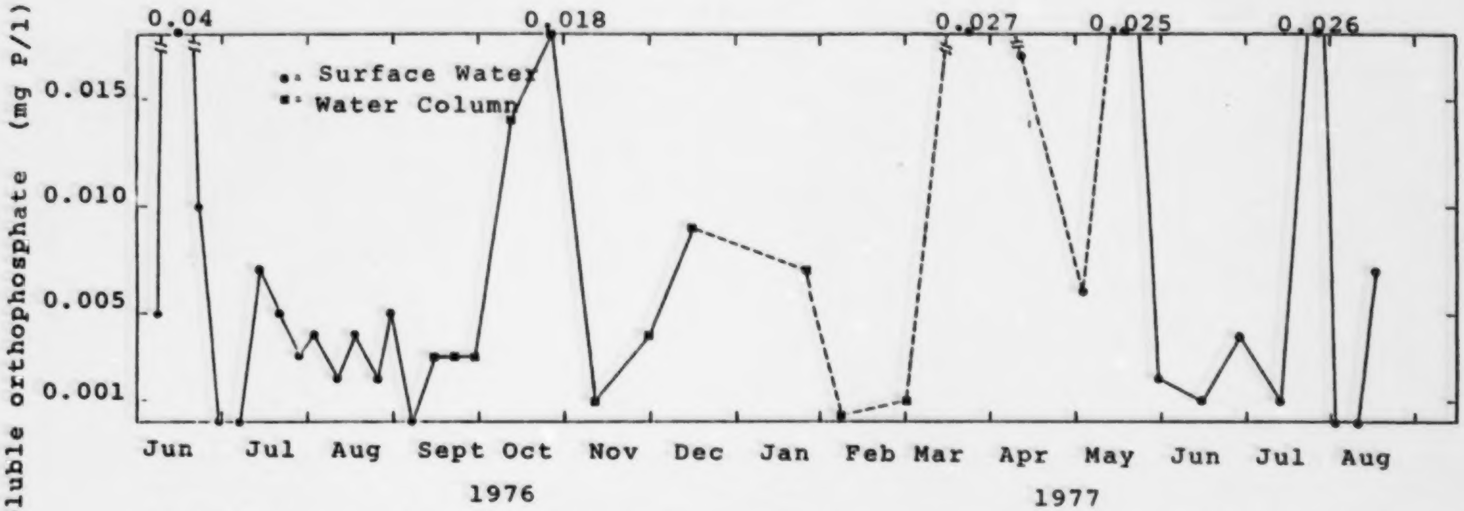


Figure 59. Average soluble orthophosphate concentration - Muddy Creek Arm

March 1977. The August peak could be due to P incorporated in algal cells since the algal numbers were high at this time and the soluble ortho P concentration was low. During October, there was an increase in the total P concentration to 1.4 mg-P/l. The soluble ortho P also increased greatly at this time. This occurred immediately after the lake became isothermal so it was most likely due to recycling of P from the sediments. The March peak of 0.21 mg-P/l was most likely due to the large input of P from Muddy Creek during February and March.

The soluble ortho P concentration fluctuated over the study period. The soluble ortho P concentration and the total number of alage were not inversely related, so soluble ortho P was most likely not limiting algal growth. This could be due to the relatively large amount of P entering from Muddy Creek which receives domestic wastewaters from the City of Wylie. There was a peak of 0.04 mg-P/l during June 1976, which was most likely due to the large tributary input of P during the spring. The peak of 0.018 mg-P/l during October was likely due to recycling from the sediments after the lake became isothermal at the end of September. During February, March, and April 1977, a large amount of P entered from Muddy Creek Arm which produced peak concentrations of 0.027 mg-P/l during March and 0.025 mg-P/l during May. There was a peak of 0.026 mg-P/l during July which was likely due to resuspension of the sediments with release of soluble ortho P to the water column since there was not a significant amount entering from Muddy Creek at this time.

Nitrogen

The average total N concentrations and the average inorganic N concentrations in Muddy Creek Arm are shown in Figures 60 and 61. The total N concentration fluctuated from 0.8 to 1.2 mg-N/l with peaks during June and November 1976 and March and April 1977. The peak of 3.7 mg-N/l during June was likely due to the large N input from Muddy Creek during May. The November peak of 2.8 mg-N/l was likely due to recycling from the sediments since there was not a large tributary input at this time.

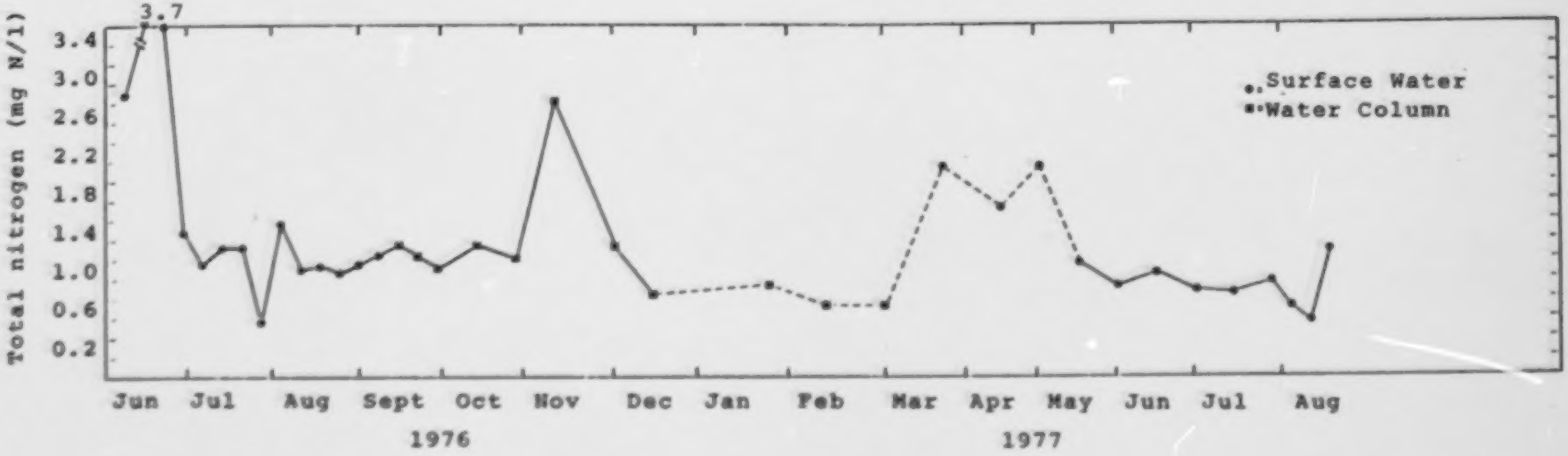


Figure 60. Average total nitrogen - Muddy Creek Arm

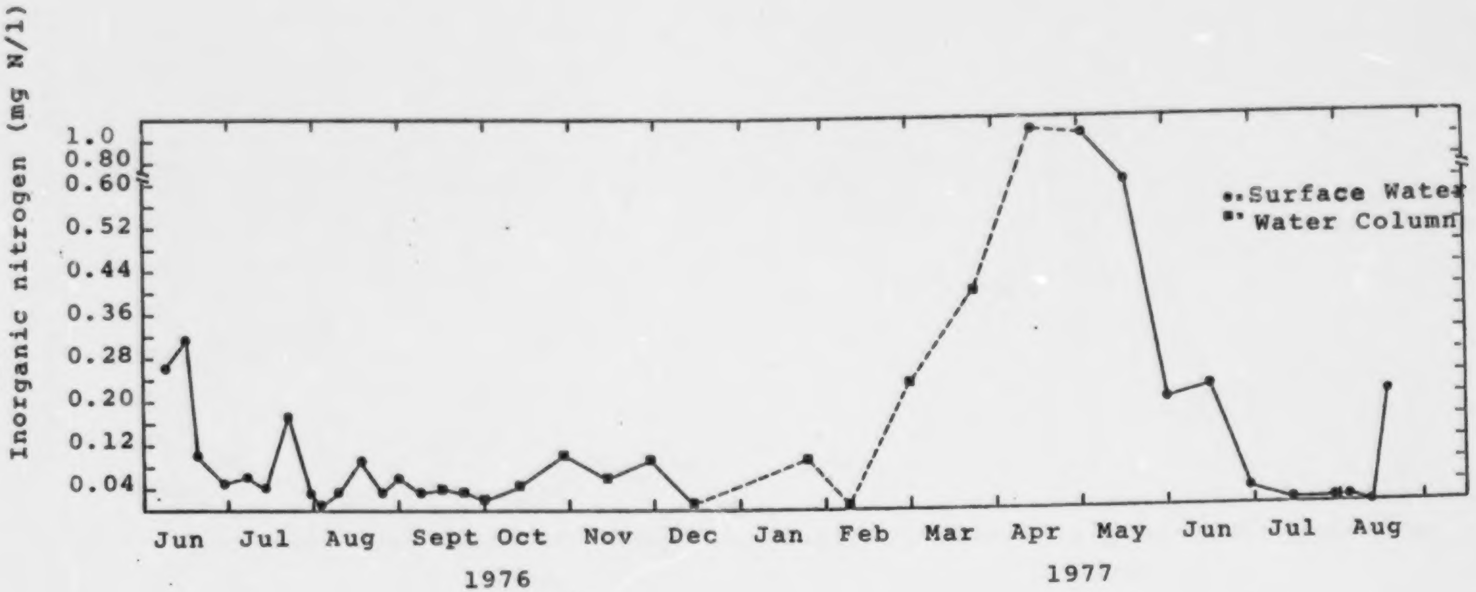


Figure 61. Average inorganic nitrogen concentration - Muddy Creek Arm

From the end of March to the beginning of May, the concentration was near 2.0 mg-N/l due to the large influx of N from Muddy Creek during these months.

The inorganic N concentration fluctuated from less than the detection limit to 1.1 mg-N/l. The concentration was generally low during the summer months due to the uptake by algae, and high during the spring months due to the large tributary input at this time. The inorganic N and soluble ortho P concentrations were below the critical concentrations for limiting algal growth during much of the study period; therefore, either could be limiting algal growth.

pH

The average pH at Stations 11 and 12 in Muddy Creek Arm were calculated and are presented in Table 19 . The table shows the pH values at the two stations were essentially equal. Figure 62 shows the pH in the surface and bottom waters at Station 11. During the summer months, the surface pH was generally slightly higher than the bottom pH, most likely due to the photosynthetic uptake of CO_2 in the surface waters and respiration of bacteria in the bottom waters. The pH ranged from 7.7 to 8.7 during the summer of 1976 with the surface pH generally being near 8.3. During the summer of 1977, the pH ranged from 8.0 to 8.8 with the surface pH generally near 8.5. The higher pH values found during the summer of 1977 were likely due to the lower solubility of CO_2 due to the higher surface water temperatures. During the winter months, the pH ranged from 7.7 to 9.0 and was generally near 8.5.

Specific Conductance

The specific conductance of the surface and bottom waters at Station 11 is shown in Figure 62. During the periods of thermal stratification, the specific conductance of the bottom waters was slightly higher than that of the surface waters. During the summer of 1976, the specific conductance in the

TABLE 19. AVERAGE pH - MUDDY CREEK ARM

	Station	
Depth	11	12
6/76 to 9/76		
S	8.1	8.0
B	8.0	8.0
10/76 to 5/77		
S	8.3	8.1
B	8.2	8.1
6/77 to 8/77		
S	8.5	8.6
B	8.4	8.4

S = Surface (1 m)
B = Bottom (0.5 m from sediments)

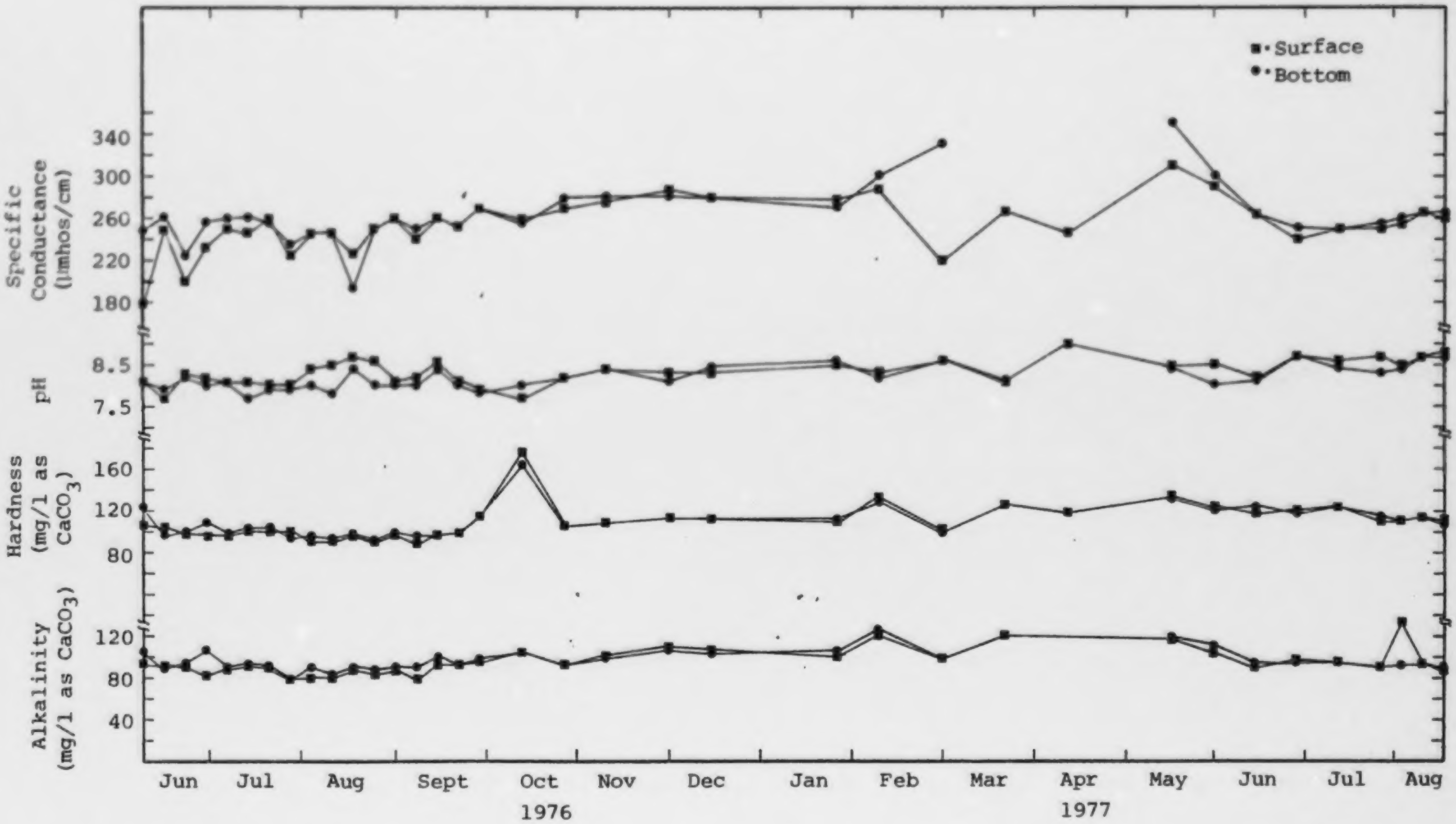


Figure 62. Chemical parameters - Station 11

bottom waters ranged from 190 to 260 µmhos/cm and was generally near 250 µmhos/cm. In the surface waters, the range was 200 to 260 µmhos/cm and was generally near 250 µmhos/cm. These values are similar to those found in the Main Body at this time. During the summer of 1977, the range was 250 to 350 µmhos/cm in the bottom waters and 240 to 310 µmhos/cm in the surface waters. The higher bottom water values were likely due to $CaCO_3$ dissolution from the sediments and precipitation in the surface waters. During the winter months, the specific conductance of the surface waters ranged from 220 to 310 µmhos/cm. The specific conductance of the bottom waters ranged from 270 to 300 µmhos/cm from October to February. There was an increase during February to 330 µmhos/cm on March 1, 1977; on May 17, 1977 the specific conductance of the bottom waters was 350 µmhos/cm. No bottom water data were collected between these two dates so it is not known if the bottom water specific conductance remained high all winter. In general, the specific conductance in Muddy Creek Arm was similar to the specific conductance in the Main Body.

Hardness

The total hardness in the surface and bottom waters at Station 11 is shown in Figure 62. The hardness of the surface and bottom waters was essentially equal throughout the study period. During the summer of 1976, the hardness ranged from 95 to 120 mg/l as $CaCO_3$ but was generally near 100 mg/l as $CaCO_3$. During the summer of 1977, the hardness ranged from 105 to 135 mg/l as $CaCO_3$. These values are similar to those found in the surface waters of the Main Body during the summer months. During the winter months, the hardness of the surface and bottom waters at Station 11 ranged from 100 to 175 mg/l as $CaCO_3$ but was generally near 120 mg/l as $CaCO_3$. This was comparable to the hardness of the Main Body waters during the winter months.

Alkalinity

The alkalinities of the surface and bottom waters were essentially equal throughout the study period as shown in Figure

62 . During the summer of 1976, the alkalinity ranged from 80 to 105 mg/l as $CaCO_3$ and during the summer of 1977, the alkalinity ranged from 80 to 135 mg/l as $CaCO_3$. These values were slightly higher than those found in the Main Body during the summer months. During the winter months, the alkalinity ranged from 90 to 115 mg/l as $CaCO_3$ which was comparable to the alkalinity in the Main Body during the isothermal period.

Secchi Depth

The average Secchi depth and average surface turbidity were highly variable in Muddy Creek Arm as shown in Figure 63 . The Secchi depth ranged from 0.1 to 0.9 m with no apparent trend over the study period. The turbidity ranged from 2 to 47 FTU's and was inversely related to the Secchi depth. Since this arm of the lake is relatively shallow (less than 5 m), the variability in the Secchi depth and turbidity could be related to the wind and organism-induced suspension of the sediments.

Light Penetration

The light penetration at 1 m at Station 11 ranged from 4 percent during early September 1976 to 18 percent during July 1976, with an average of 12 percent. This was comparable to the transparency of the water in the East Fork Arm.

The compensation depth at Station 11 was typically at 3 m and varied between 1 and 4 m during the study period. The relationship between the compensation depth and the Secchi depth was found to be:

$$1\%\ D = 3.6 \times \text{Secchi Depth}$$

This is similar to the relationship found in Rowlett Arm.

The extinction coefficient ranged from 1.2 m^{-1} to 2.5 m^{-1} as shown in Figure 64. The average value during the study period was 1.5 m^{-1}. The relationship between the extinction coefficient and the Secchi depth is described by the following equation:

$$Ke = \frac{1.4}{(\text{Secchi})^{0.6}}$$

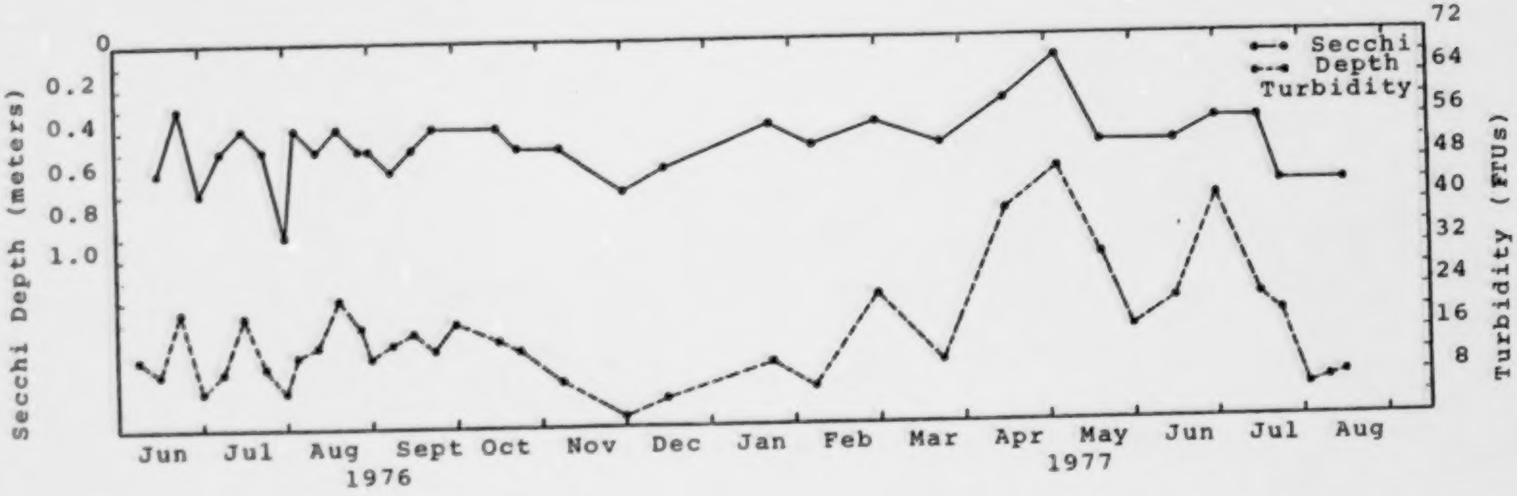


Figure 63. Average Secchi depth and turbidity - Muddy Creek Arm

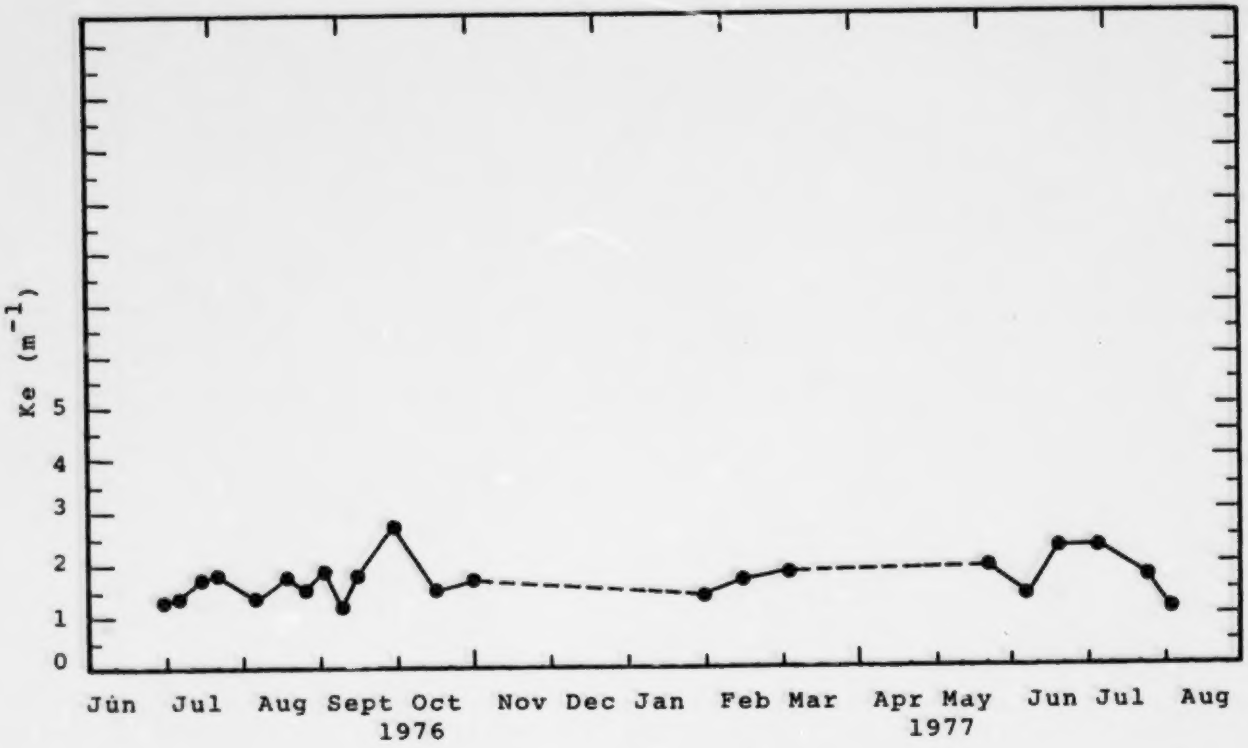


Figure 64. Extinction coefficient - Station 11

The relationship between the uncorrected chlorophyll a concentration and the extinction coefficient was found to be:

$$Ke = 1.2 + 0.07 \{\text{uncorrected chlorophyll } \underline{a}\ (\mu g/l)\}$$

The extinction coefficient due to factors other than the uncorrected chlorophyll a concentration (1.2) was intermediate between the Main Body and Rowlett Arm values.

Algae

The algal population of Muddy Creek Arm showed the same cycle as the Main Body population with peaks during both summers and a peak in January. The average algal numbers in Muddy Creek Arm are shown in Figure 65. The total number increased to 19,500 organisms/ml in June 1976, with the increase mainly due to the green alga, *Chlorella*. This increase was likely due to the large concentrations of soluble ortho P and inorganic N available for algal growth at this time. The total number remained high through August, but the composition of the population changed. As the summer progressed, the blue-green alga, *Oscillatoria*, became dominant and the green algae decreased in numbers.

The algal population started increasing again in December and reached a peak of 35,000 organisms/ml at the end of January. This peak was due to a bloom of the diatoms, *Cyclotella* and *Fragillaria*. The nutrients responsible for this growth were most likely recycled from the sediments since there was not a large input from Muddy Creek at this time.

The algal population increased from June to August 1977 with a peak of 27,700 organisms/ml in August. The blue-green alga, *Oscillatoria*, and the green alga, *Chlorella*, were both present in high numbers at this time. The concentrations of soluble ortho P and inorganic N were both high before the bloom but were depleted by the algae during the bloom.

In Muddy Creek Arm, the algal population was higher than in the Main Body and ranged from 4,000 to 35,000 organisms/ml with an average concentration of approximately 12,000 organisms/ml during non-bloom periods.

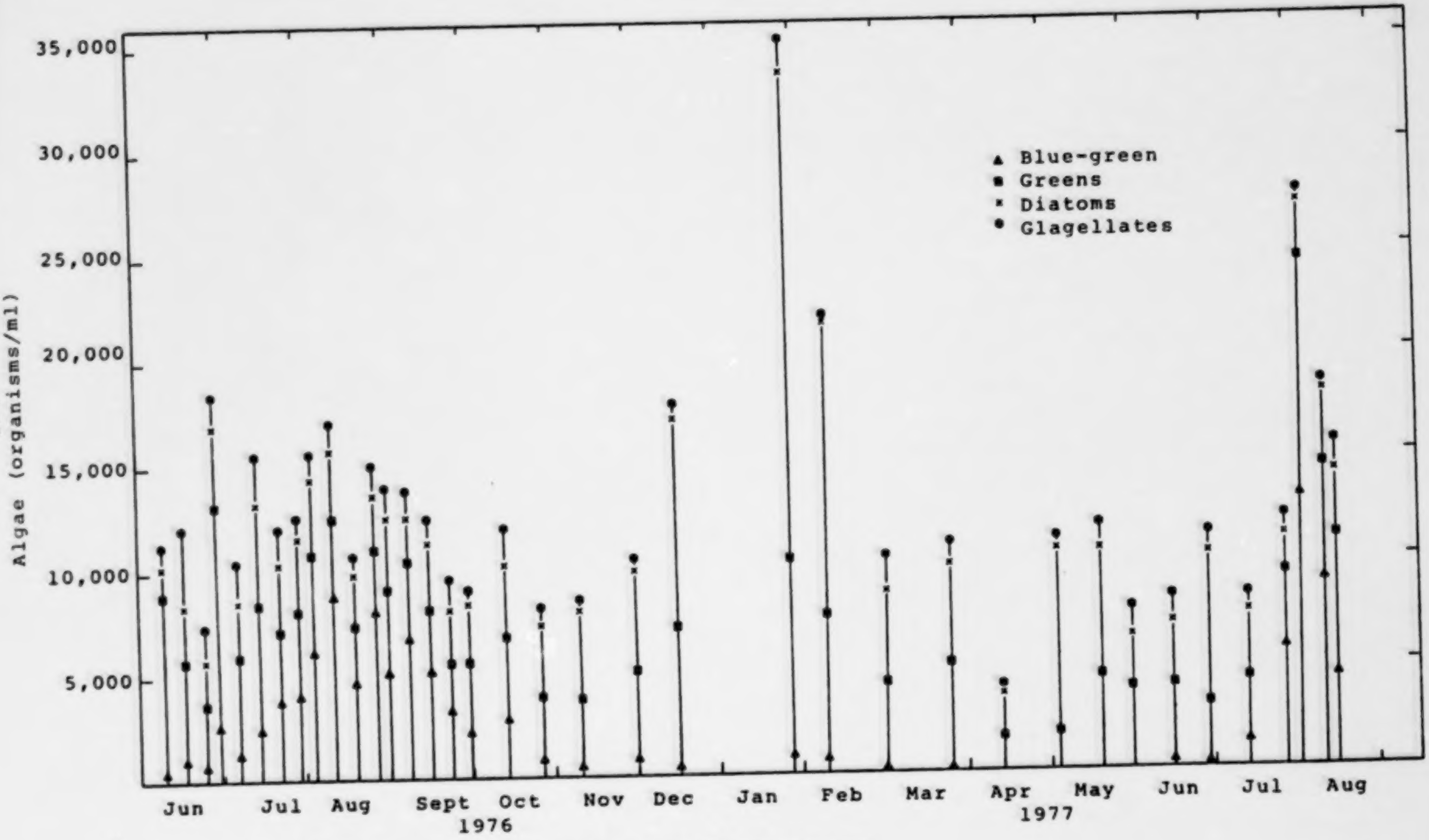


Figure 65. Algal numbers and types - Muddy Creek Arm

Chlorophyll a

The average uncorrected chlorophyll a concentrations in Muddy Creek Arm are shown in Figure 66. There were large fluctuations from week to week, but there appeared to be a trend similar to that found in the Main Body in which there were peaks in the summer and winter. In Muddy Creek Arm, the uncorrected chlorophyll a ranged from 7 to 72 µg/l with peaks of 36 µg/l in August 1976, 38 µg/l in January 1977, and 72 µg/l in June 1977. These concentrations were much higher than those found in the Main Body. The chlorophyll a concentration followed the same trend as the uncorrected chlorophyll a and ranged from 6 to 72 µg/l. The pheophytin a concentration ranged from 0 to 27 µg/l with the largest concentrations found during the winter months.

Primary Productivity

The primary productivity of Muddy Creek Arm was highly variable during the year, ranging from 128 to 562 mg $C/m^2/hr$ as shown in Table 13. The highest productivities were found during the summer months, and the lowest were found during the fall and spring. The euphotic zone varied from 1.5 to 3.5 m and was likely influenced by the amount of inorganic turbidity in the water due to the wind and organism-induced suspension of the sediments.

Threshold Odor Number

Figure 67 shows the average threshold odor numbers in Muddy Creek Arm. During the summer of 1976, the T.O.N. ranged from 6 to 12 with musty, grassy, rooty, and, occasionally septic odor characteristics. No T.O.N. samples were collected in Muddy Creek Arm during the winter months. During the summer of 1977, the T.O.N. ranged from 7 to 14 with vegetable, musty, and grassy odor characteristics. In general, the T.O.N.'s in Muddy Creek Arm were slightly higher than those found in the Main Body during the summer months.

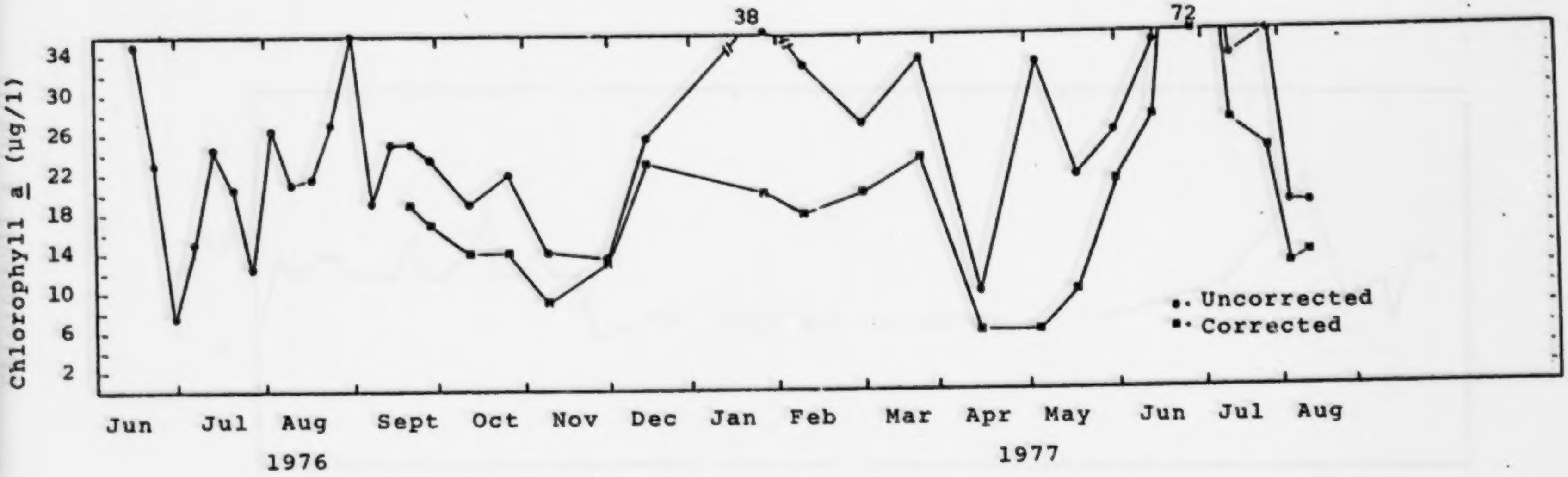


Figure 66. Chlorophyll a concentration - Muddy Creek Arm

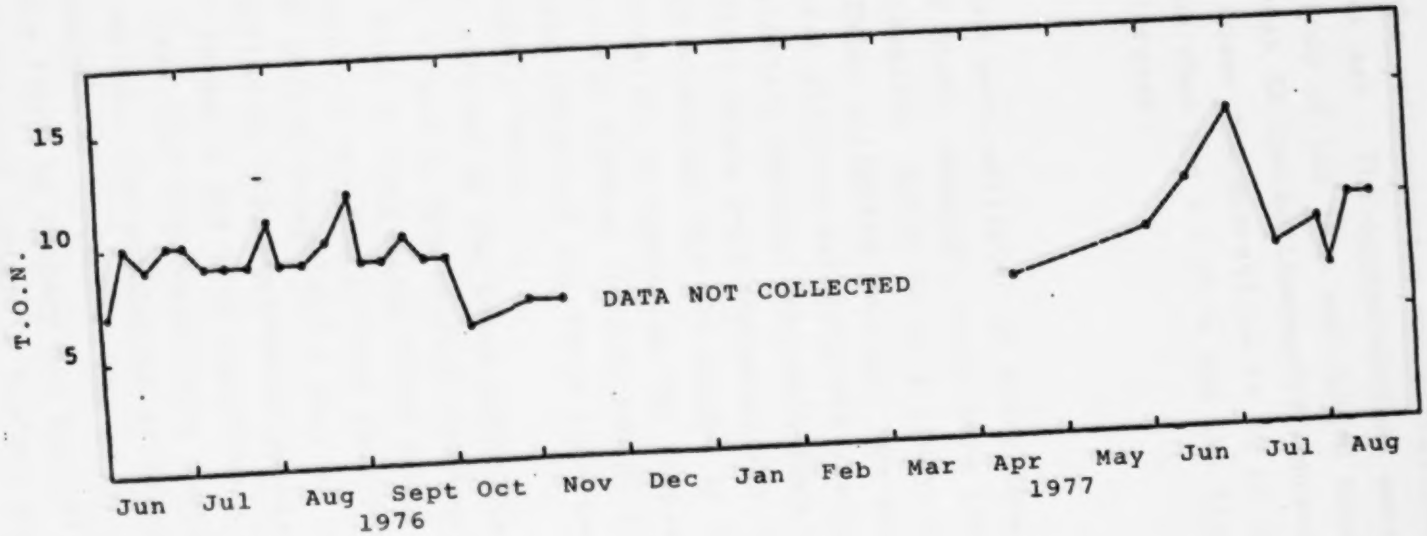


Figure 67. Threshold odor number - Muddy Creek Arm

Actinomycetes

Figure 68 presents the actinomycetes concentrations at Station 12 in Muddy Creek Arm. The concentrations were higher than those in the Main Body of the lake and ranged from 4 to 47 organisms/ml. The peak in the actinomycetes concentration occurred in April with lower concentrations (4 to 15 organisms/ml) during the summer months when the T.O.N.'s and the blue-green algal population were largest.

MISCELLANEOUS RESULTS

Zooplankton Data

Zooplankton samples were collected at every lake sampling station during the UTD study; however, there were insufficient funds to analyze the samples. Prior to this study, the Texas Water Quality Board (TWQB) collected vertical tow zooplankton samples with a 76 μ mesh plankton net from six stations on Lake Ray Hubbard at approximately monthly intervals from June 1974 to September 1975. While there were approximately one to two years between the TWQB study and the UTD study, it is felt that it would be informative to summarize the zooplankton data obtained by the TWQB. For further information on this subject, contact the Duncanville Office of the Texas Department of Water Resources (formerly TWQB).

Zooplankton was sampled by the Texas Water Quality Board at Stations 1, 2, 3, 7, and 9, and halfway between Stations 11 and 12 where Miller Road crosses Muddy Creek Arm. At Station 1 the zooplankton population was small (less than 700 organisms/l) throughout the study period except for a peak of 4400 organisms/l in May 1975. At Station 2, the zooplankton population peaked at 1200 organisms/l in April and 1200 organisms/l in May 1975. The population was less than 600 organisms/l throughout the rest of the study period. The population at Station 7 was small (less than 800 organisms/l) throughout the study period. The largest numbers were found in January and May 1975. The zooplankton population was much larger in Rowlett Arm, ranging

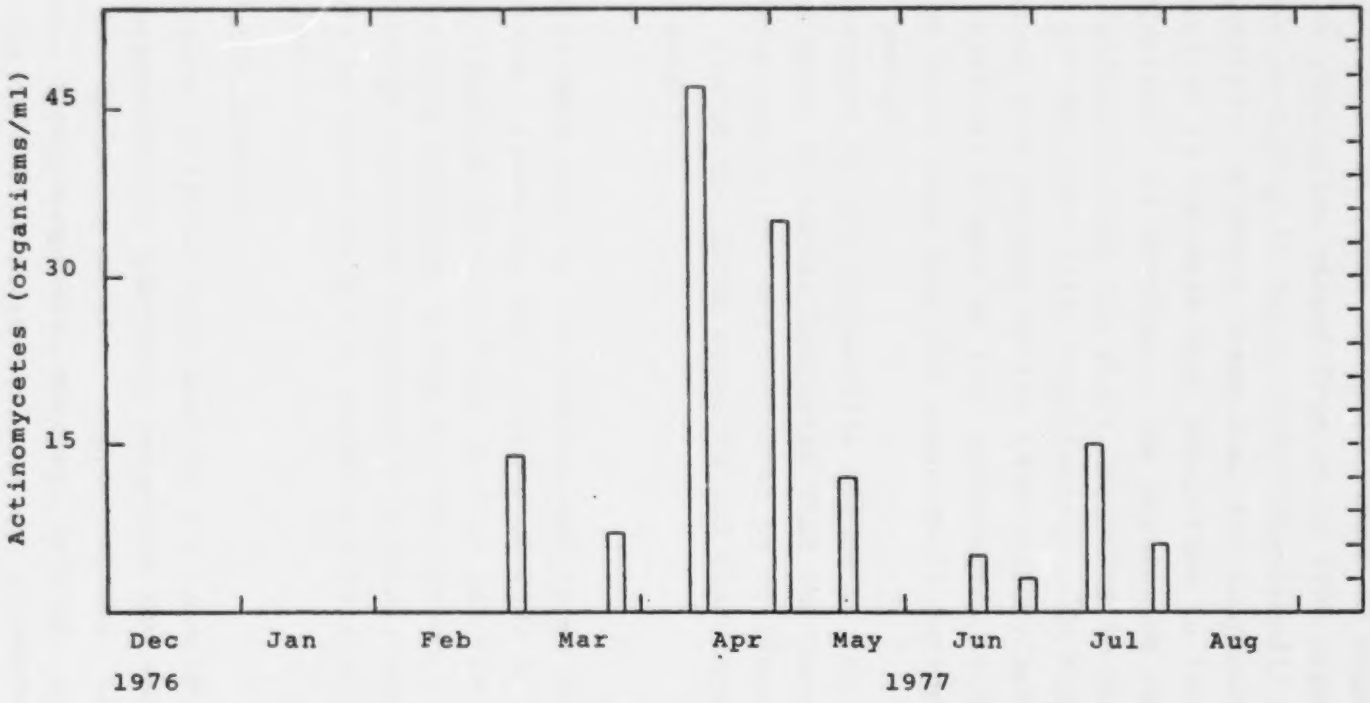


Figure 68. Actinomycetes - Station 12

from 16 to 11,500 organisms/l at Station 9. The peak concentrations were found in June 1974 (2200 organisms/l), in May 1975 (11,500 organisms/l),and June 1975 (1300 organisms/l). In the East Fork Arm, the zooplankton population was similar to the Main Body population in terms of total number of organisms. At Station 3, the population ranged from 24 to 2000 organisms/l with the peaks occurring in March (2000 organisms/l) and May 1975 (1500 organisms/l). In Muddy Creek Arm, the zooplankton population was similar to the Main Body population in terms of total number of organisms. At Station 3, the population ranged from 24 to 2000 organisms/l with the peaks occurring in March (2000 organisms/l) and May 1975 (1500 organisms/l). In Muddy Creek Arm, zooplankton were sampled by the TWQB halfway between Stations 11 and 12. There was a peak of 2000 organisms/l in May 1975 with the population being less than 500 organisms/l during the rest of the study period.

Mr. J. Pettitt of the Duncanville Office of the Texas Department of Water Resources indicated that the Lake Ray Hubbard zooplankton population was dominated by rotifers and other small forms. Few of the large copepods and cladocerans were found in the samples.

Fish

No studies were done on the numbers and types of fish in Lake Ray Hubbard. Potential significance of fish in altering nutrient load-response relationships in this lake is unknown. There is increasing evidence in the eutrophication literature that fish, through selective zooplankton grazing, may play an important role in determining the number and type of phytoplankton in lakes.

Heavy Metals and Sulfate

Samples were collected from each of the lake and tributary stations at approximately quarterly intervals for analysis of sulfate and the following metals: copper, nickel, zinc, cadmium, chromium, lead, iron, manganese, mercury, calcium, sodium, and potassium. The data from these analyses are presented in the appendix to this report. Review of these data indicates that some concentrations of heavy metals in Lake Ray Hubbard water may be sufficiently high to be adverse to algal growth in a low alkalinity and hardness water; however, these concentrations are not expected to be toxic to algae in Lake Ray Hubbard water. It is unlikely that any of the heavy metals or other constituents measured would have a significant adverse effect on nutrient load-response relationships in Lake Ray Hubbard.

NUTRIENT LOADS*

The nutrient loads to Lake Ray Hubbard from the tributaries were calculated from measured concentrations and estimated flows. DWU provided estimated flows for the small tributaries (Muddy, Rush, Cottonwood, Camp, Squabble, and Yankee Creeks); USGS provided flow data for Rowlett Creek and the East Fork of the Trinity River below Lake Lavon; and the flows at Station 24 were from measurements made at the Garland-Rowlett sewage treatment plant. DWU provided information on the amount of water taken at the Forney Pump Station (Station 13) and the amount of water released at the Lake Ray Hubbard Spillway (Station 22).

Nutrient loads were calculated on a weekly basis since samples were collected for nutrient analyses once a week at the beginning of the study in the summer of 1976. The sampling frequency was changed to once every two to three weeks for the fall of 1976 and winter, spring, and summer of 1977. The daily flow data were averaged to obtain weekly flows with the sampling date as the midpoint of the week. The concentrations of nutrients found on the sampling date were multiplied by the average flow to get the loadings. During the periods of time when the samples were collected at less than weekly intervals, the average concentration of nutrients between adjacent sampling dates was used to estimate the loads. During monitored periods of high flow, when more than one sample was collected in a day, the concentrations of the samples obtained were averaged. The average concentration was multiplied by the total flow that occurred during the high flow period. The concentrations were averaged since no relationship was found between flow and concentrations of nutrients. With the return of more normal flows to the streams, the loadings were calculated using the average weekly flow.

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*Michael Gary contributed to this section of the report.

The total P loads at Station 15 during February, March, and April 1977 were calculated using daily flows and average concentrations between sampling dates. These loads were essentially equal to the loads based on average weekly flows. Therefore, it appears reasonable to estimate nutrient loads based on weekly flows.

All nutrient load estimates made for Lake Ray Hubbard ignore any nutrient input that occurs from land runoff to the tributaries below the sampling point. The largest error associated with this approach would be for Rowlett Creek below the Highway 78 bridge. The estimated nutrient loads to the lake also ignore any runoff from the land directly to the lake. It is felt that the errors introduced by neglecting these sources are small based on the relative land areas involved. DWU has estimated that the nutrient loads presented in this report represent 77 percent of the total land area in the Lake Ray Hubbard watershed.

The amounts of nutrients transported to Lake Ray Hubbard in each tributary and the amounts leaving the lake at the spillway and the East Side Water Treatment Plant intake during each month of the study are presented in Tables 20 to 31. Table 20 presents the estimated amounts of nutrients removed from Lake Ray Hubbard at the East Side Water Treatment Plant intake as a result of the City of Dallas' using Lake Ray Hubbard as a domestic water supply. The values during August 1976 are less than what would normally be expected, due to the fact that the City of Dallas stopped using Lake Ray Hubbard as a domestic water supply during parts of July and August. Table 21 presents the amounts of nutrients removed from Lake Ray Hubbard in the water released from the lake at the spillway to the East Fork of the Trinity River.

Table 22 presents the data for the estimated nutrient transport of Rowlett Creek at Miller Road. Since this station was in the backwater of Lake Ray Hubbard, it was not suitable for calculating the nutrient loads to the lake from Rowlett

TABLE 20. NUTRIENT OUTFLOW FROM THE EAST SIDE WATER TREATMENT PLANT INTAKE - STATION 13

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	12	35	64	44	6	114	570
June	24	43	202	13	9	224	2995
July	135	183	92	78	0	170	1193
Aug	5	78	29	3	1	33	975
Sept	43	103	43	99	5	147	1426
Oct	54	252	223	438	79	740	2744
Nov	20	84	115	67	38	220	1506
Dec	18	259	185	232	0	417	2006
1977							
Jan	32	276	100	1742	10	1852	3412
Feb	36	1055	105	841	1	947	3426
Mar	94	3584	184	655	87	926	8344
Apr	56	273	0	1698	115	1813	1914
May	37	289	1155	3000	192	4347	2323
June	361	605	0	1042	102	1044	3183
July	34	332	0	0	0	0	4980

*May 16-31

Units are kg as N or P.

TABLE 21. NUTRIENT OUTFLOW - STATION 22: LAKE RAY HUBBARD SPILLWAY

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	4	21	56	21	4	81	486
June	49	85	115	83	8	206	2325
July	25	151	423	11	0	434	3627
Aug	13	34	36	17	2	55	561
Sept	5	37	32	13	0	45	764
Oct	35	133	73	30	11	114	578
Nov	10	33	104	19	0	123	698
Dec	12	49	49	55	8	112	796
1977							
Jan	1	25	17	38	2	57	380
Feb	9	146	15	91	4	110	603
Mar	1522	22706	1602	8614	1330	11546	95224
Apr	228	2219	0	10489	795	11284	27198
May	12	155	0	2258	119	2377	1646
June	2	18	2	117	12	131	225
July	3	10	0	2	0	2	174

*May 16-31

Units are kg as N or P.

TABLE 22. NUTRIENT INFLOW - STATION 14: ROWLETT CREEK AT MILLER ROAD

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	1186	2588	2583	2941	205	5729	5008
June	7570	8611	5034	6660	492	12186	20040
July	2909	3583	2585	4286	372	7243	5329
Aug	1811	1975	224	883	392	1499	1126
Sept	3205	4464	1524	3820	446	5790	5814
Oct	6455	5338	6099	11515	2970	2058	3908
Nov	3598	4191	3407	2358	755	6520	2172
Dec	3863	5142	806	4974	242	6022	8489
1977							
Jan	5166	5995	4694	3530	397	8621	5793
Feb	3803	54595	2819	12085	878	15782	67502
Mar	19049	47945	646	26636	3162	30444	39551
Apr	14795	16507	366	30461	2057	32885	39709
May	3410	4281	2014	3572	464	6050	2846
June	3534	4286	4598	4524	905	10027	3393
July	1937	2413	1742	1278	252	3272	1646

*May 16-31

Units are kg as N or P.

Note: This station is in backwater of Lake Ray Hubbard and is not a reliable estimate of nutrient loadings to the lake.

Creek. This station receives nutrients from land runoff and the Plano and Garland-Rowlett sewage treatment plants. It should be noted that only part of the domestic wastewaters from the City of Garland are treated at the Garland-Rowlett plant. The remainder are treated at the Duck Creek plant which discharges into the East Fork of the Trinity River below Lake Ray Hubbard. Due to the backwater conditions that existed at Station 14, it was decided that a more reliable method for estimating the nutrient loads to Lake Ray Hubbard via Rowlett Creek would be to sum the loads at Rowlett Creek at Highway 78 (Station 15) and the loads from the Garland sewage treatment plant (Station 24). Table 23 presents the amounts of nutrients transported by Rowlett Creek at Highway 78 (Station 15). This station was located above the point where the Garland sewage treatment plant discharges to Rowlett Creek so the nutrient loads were derived from nutrient runoff from the land and the Plano sewage treatment plant. Table 24 presents the amounts of nutrients discharged from the Garland-Rowlett sewage treatment plant. The samples were taken near the point at which the effluent enters Mills Branch which flows a short distance and enters Rowlett Creek just above Station 14.

Table 25 presents the estimated nutrient loads to Lake Ray Hubbard from Muddy Creek. The nutrients derived from the Wylie sewage treatment plant and land runoff to Muddy Creek were measured at this station. The nutrient load contributions from Cottonwood and Rush Creeks are presented in Tables 26 and 27 . Land runoff was the only source of nutrients for these two small tributaries. Table 28 presents the nutrient loads from the East Fork of the Trinity River. The sampling station was located just downstream from the spillway of Lake Lavon. Most of the values presented in this table are zeros since there was no flow in the river since no water was released from Lake Lavon during the study period. The nutrient loads from Camp Creek to Lake Ray Hubbard are presented in Table 29 . The estimated loadings given in this table are not considered to be highly reliable

TABLE 23 . NUTRIENT INFLOW - STATION 15: ROWLETT CREEK AT HWY 78

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	339	2636	1152	569	145	1866	9900
June	2491	3136	2851	3811	185	6847	9751
July	895	2003	2224	2286	199	4709	5063
Aug	220	332	444	153	50	646	650
Sept	714	5419	89	2047	209	2345	8607
Oct	1006	1263	1873	4692	1208	7773	1892
Nov	376	495	1315	919	233	2467	782
Dec	1006	4074	1980	3331	205	5516	6634
1977							
Jan	3662	4497	5320	3834	493	9647	4700
Feb	2329	33118	2580	14015	802	17397	51907
Mar	7014	50748	427	23237	3212	26876	29914
Apr	6198	28947	0	28424	2164	30588	26821
May	1467	2215	3167	2720	1017	6904	1930
June	1318	1717	6148	1926	561	8635	1800
July	191	345	1684	574	118	2376	571

*May 16-31

Units are kg as N or P.

TABLE 24 . NUTRIENT INFLOW - STATION 24: GARLAND SEWAGE TREATMENT PLANT

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	916	1895	1208	231	66	1505	1091
June	2811	3153	1579	1125	50	2754	2521
July	3107	3401	1554	1108	134	2796	1978
Aug	3503	3470	2438	1034	37	3509	1125
Sept	3196	2748	1172	2174	45	3391	1704
Oct	3202	4214	4491	1804	233	6528	1809
Nov	2335	2642	1033	953	88	2074	1849
Dec	2589	3358	2980	780	73	3833	1690
1977							
Jan	2723	3358	2869	611	77	3557	987
Feb	2539	3157	3000	647	65	3712	2002
Mar	5862	6564	4952	616	158	5726	3366
Apr	3665	4553	4152	1029	69	5250	3397
May	3357	3916	3109	1085	81	4275	1622
June	2702	3123	1153	1188	124	2465	1102
July	2526	3001	1535	1018	35	2588	978

*May 16-31

Units are kg as N or P.

TABLE 25. NUTRIENT INFLOW - STATION 16: MUDDY CREEK AT PLEASANT VALLEY ROAD

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May	389	1661	458	1177	64	1699	15796
June	378	862	196	286	33	515	3511
July	133	326	39	313	12	364	1028
Aug	0	0	0	0	0	0	0
Sept	5	35	8	1	0	9	83
Oct	0	0	0	0	0	0	0
Nov	8	83	9	4	2	15	144
Dec	80	458	20	29	20	69	556
1977							
Jan	410	523	20	369	47	436	537
Feb	516	2081	56	2048	79	2182	3325
Mar	1044	15823	164	7488	469	8121	12980
Apr	936	5422	0	9061	2718	11779	6115
May	76	325	79	580	417	1076	1212
June	11	134	0	0	0	0	315
July	0	0	0	0	0	0	0

Units are kg as N or P.

TABLE 26. NUTRIENT INFLOW - STATION 17: COTTONWOOD CREEK AT STONEWALL ROAD

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	12	29	8	6	2	16	119
June	12	18	2	5	1	8	61
July	13	27	4	24	1	29	94
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	2	3	1	3	0	4	8
Dec	16	22	3	34	1	38	63
1977							
Jan	10	13	2	26	1	29	36
Feb	19	118	2	150	4	156	168
Mar	122	1643	24	521	27	572	1216
Apr	13	26	0	196	4	200	106
May	4	8	0	126	2	128	43
June	0	0	0	0	0	0	0
July	0	0	0	0	0	0	0

*May 16-31

Units are kg as N or P.

TABLE 27. NUTRIENT INFLOW - STATION 18: RUSH CREEK AT KRAMOR LANE

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	3	23	10	6	1	17	13
June	7	16	8	7	1	16	179
July	2	20	5	47	3	55	117
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	0	1	1	2	0	3	7
Dec	2	5	4	13	4	21	52
1977							
Jan	1	2	1	19	4	24	18
Feb	6	79	2	109	3	114	164
Mar	31	1161	26	979	5	1010	893
Apr	4	15	5	27	2	34	86
May	1	8	13	9	2	24	58
June	1	12	1	2	1	4	10
July	0	2	0	1	0	1	5

*May 16-31

Units are kg as N or P.

TABLE 28. NUTRIENT INFLOW - STATION 19: EAST FORK-TRINITY RIVER AT HWY 78

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	0	1	0	0	0	0	3
June	0	0	0	0	0	0	1
July	0	1	0	0	0	1	5
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	0	0	0	0	0	0	0
Dec	0	0	0	0	0	0	0
1977							
Jan	0	0	0	0	0	0	0
Feb	0	1	0	1	0	1	2
Mar	0	0	0	0	0	0	0
Apr	0	0	0	0	0	0	0
May	0	0	0	0	0	0	0
June	0	0	0	0	0	0	0
July	0	0	0	0	0	0	0

*May 16-31

Units are kg as N or P.

TABLE 29. NUTRIENT INFLOW - STATION 20: CAMP CREEK AT HWY 205

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May	41	493	147	94	27	268	2358
June	29	106	61	35	7	103	1319
July	5	132	25	57	4	86	699
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	0	3	1	6	0	7	15
Dec	0	0	0	0	0	0	0
1977							
Jan	1	14	7	47	0	54	79
Feb	21	645	11	493	8	512	968
Mar	394	15316	262	3581	239	4082	10622
Apr	59	622	0	901	74	975	1350
May	6	46	101	180	13	294	359
June	1	18	0	0	0	0	95
July	0	4	1	0	0	1	60

Units are kg as N or P.

since the sampling station was located in backwater of Lake Ray Hubbard. There was no other easily accessible point from which this tributary could be sampled. The nutrients transported in this creek were derived from land runoff.

Table 30 presents the nutrient inflow from Squabble Creek based on measurements at Highway 205. The City of Rockwall discharges its treated domestic wastewaters between the sampling location and Lake Ray Hubbard. Since measured values were not available for the amounts of nutrients contributed to the lake from the Rockwall sewage treatment plant, it was necessary to estimate these based on population equivalents for flow and typical nutrient content of domestic wastewaters. It was assumed that the flow for the Rockwall sewage treatment plant was 1.5 X 10^6 l/day (NCTCOG, 1977a). Based on a population of 4700 people, this gives a value of 319 l/capita/day. It was assumed that the soluble ortho P content of the untreated domestic wastewaters was 1.4 kg/capita/yr; the total P content was 2 kg/capita/yr; the ammonia content was 2.7 kg/capita/yr as N; the total N content was 3.6 kg/capita/yr; and there was no nitrate in the untreated domestic wastewaters. These assumptions are based on a review of the literature by Rast and Lee (1977). In order to estimate the load to the lake, it was assumed that the Rockwall plant would remove 20 percent of the particulate phosphorus and 25 percent of the nitrogen as part of the treatment process. This results in an estimated soluble orthophosphate load from the Rockwall sewage treatment plant of 548 kg/month, a total P load of 736 kg/month, and a total nitrogen load of 1058 kg/month. A comparison between these values and the values presented in Table 30 shows that the Rockwall sewage treatment plant was the primary nutrient source for Squabble Creek.

It should be noted that different per capita loading figures from domestic wastewaters are used in this report for the different wastewater treatment plants in the Lake Ray Hubbard basin. These differences reflect the authors' judgement

TABLE 30. NUTRIENT INFLOW - STATION 21: SQUABBLE CREEK AT HWY 205*

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May	1	86	82	15	8	105	680
June	6	15	8	7	1	16	140
July	1	14	3	7	0	10	113
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	0	0	0	0	0	0	0
Dec	1	2	0	0	0	0	9
1977							
Jan	0	1	1	1	0	2	7
Feb	4	15	1	13	0	14	23
Mar	90	1548	28	1076	27	1131	1713
Apr	13	45	0	29	9	38	248
May	0	7	6	4	1	11	50
June	0	0	0	0	0	0	0
July	0	0	0	0	0	0	0

*These values do not include the contributions from the Rockwall sewage treatment plant. See Table 34 for an estimate of the amounts of nutrients entering Squabble Creek from the Rockwall sewage treatment plant.

Units are kg as N or P.

on the amounts of nutrients derived from the different treatment plants. Consideration was given to the type of treatment plant, the age, and expected overall operating efficiency.

Table 31 presents the amounts of nutrients transported to Lake Ray Hubbard by Yankee Creek. These nutrients were derived from land runoff and a package sewage treatment plant which receives domestic wastewaters from a housing development.

The amounts of nutrients derived from precipitation and dustfall onto the lake surface were not measured in the tributary sampling. An attempt was made to determine the amounts of nutrients derived from the atmosphere; however, work in this area was severely constrained by the limited funding available. Precipitation and dustfall collection buckets were placed at three sites around the main body of the lake, one site in Parker, and one site in Denison, as discussed in the Analytical Methods Section of this report. The atmospheric loading coefficients calculated from this study are presented in Table 32. The data obtained from the atmospheric loading study showed a great deal of variability among the sampling sites and among samples collected at one site. The loading coefficients for soluble orthophosphate ranged from 0.003 to 0.03 $g/m^2/yr$; the total P coefficients ranged from 0.06 to 0.11 $g/m^2/yr$; and the inorganic N coefficients ranged from 0.19 to 0.71 $g/m^2/yr$. In general, these numbers are similar to those found in other parts of the US. Due to the variability of the data, it was decided to use the values reported by Rast and Lee (1977) which are based on a number of studies conducted across the US for estimating the atmospheric nutrient loadings. It was assumed that 0.1 $g/m^2/yr$ of phosphate came from the atmosphere, 0.02 $g/m^2/yr$ from rainfall, and 0.08 $g/m^2/yr$ from dry fallout. The total nitrogen contribution from the atmosphere was assumed to be 2.4 $g/m^2/yr$ which was broken down to 0.8 $g/m^2/yr$ from rainfall and 1.6 $g/m^2/yr$ from dry fallout. Based on a review of the literature and the experience of the authors, it was felt that the following distribution was appropriate. Ten percent of

TABLE 31. NUTRIENT INFLOW - STATION 23: YANKEE CREEK AT MEADOWVIEW ROAD

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	12	15	2	7	0	9	36
June	4	5	2	2	0	5	20
July	0	0	0	0	0	0	2
Aug	0	0	0	0	0	0	0
Sept	0	0	0	0	0	0	0
Oct	0	0	0	0	0	0	0
Nov	0	0	0	0	0	0	0
Dec	0	0	0	0	0	0	2
1977							
Jan	0	0	0	1	0	1	2
Feb	1	3	0	2	0	2	8
Mar	4	25	3	47	1	51	95
Apr	1	2	1	33	1	35	14
May	0	0	0	0	0	0	0
June	0	0	0	0	0	0	0
July	0	0	0	0	0	0	0

*May 16-31

Units are kg as N or P.

TABLE 32. ATMOSPHERIC LOADING COEFFICIENTS FOR NITROGEN AND PHOSPHORUS

Location	Soluble Orthophosphate			Total Phosphorus			Inorganic Nitrogen		
	$\overline{X}$	SD	n	$\overline{X}$	SD	n	$\overline{X}$	SD	n
Lake Ray Hubbard*									
Precipitation & Dry Fallout	0.03	0.02	7	-	-	-	0.68	0.27	6
Dry Fallout	-	-	-	-	-	-	0.35	0.09	6
Parker, Texas									
Precipitation & Dry Fallout	0.009	0.007	9	0.06	0.6	5	0.51	0.2	8
Dry Fallout	0.003	0.004	4	-	-	-	0.2	0.05	4
Denison, Texas									
Precipitation & Dry Fallout	0.02	0.01	4	0.11	-	1	0.71	0.07	4
Dry Fallout	-	-	-	-	-	-	0.19	-	1

Units are $g/m^2/yr$.

*Average of the three stations around the lake.

Dash (-) indicates insufficient data to calculate.

$\overline{X}$ = Mean, SD = Standard deviation, n = Number of samples.

the total P was soluble ortho P, 25 percent of the total N was organic N, 37.5 percent was NO_3^- as N, and 37.5 percent was NH_3 as N. Using these atmospheric loading coefficients and the lake surface area of 8.8×10^7 m^2, the atmospheric phosphorus contribution was calculated to be 24 kg/day with 2.4 kg/day in the form of soluble orthophosphate. The total nitrogen contribution was 579 kg/day. The total N was broken down to 145 kg/day of organic N, 217 kg/day of NO_3^- as N, and 217 kg/day of NH_3 as N.

Table 33 presents the total nitrogen and phosphorus loads to Lake Ray Hubbard from the tributaries and the atmosphere during each month of the study from May 1976 to July 1977. As expected, the total loads for each parameter were relatively constant during low flow conditions due to the essentially steady inputs from the sewage treatment plants discharging to tributaries of the lake. However, during the relatively wet period of February through April 1977, the loads of nutrients increased markedly. The total P load during dry weather conditions ranged from 4800 to 10,700 kg/month. During March 1977 over 94,000 kg of total P were transported to the lake from the tributaries and the atmosphere. Under low flow conditions, the soluble ortho P was approximately 70 percent of the total P. However, during February and March 1977, only 15 percent of the total P input was in the soluble ortho P form. This points to the importance of land runoff in the transport of particulate P to the lake. As discussed in a subsequent section of this report, only approximately 20 percent of the particulate (non-soluble ortho P) would be expected to become available to support algal growth in Lake Ray Hubbard.

Phosphorus

Table 34 presents the annual tributary and atmospheric nutrient loads to Lake Ray Hubbard and Table 35 presents the percent of the total load of each of the nutrient forms contributed by each tributary. The annual tributary load of total P was 238,341 kg P, and the annual atmospheric load was 8,676 kg P

TABLE 33. TOTAL NUTRIENT INPUT FROM TRIBUTARIES AND ATMOSPHERE

Month	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
1976							
May*	2,310	8,301	10,367	8,615	313	19,295	34,599
June	6,335	8,773	12,007	11,788	278	24,073	22,106
July	4,753	7,386	11,154	10,352	353	21,859	13,702
Aug	4,320	6,215	10,182	7,697	87	17,966	6,378
Sept	4,512	10,664	8,569	10,732	254	19,555	15,017
Oct	4,805	6,939	13,664	13,006	1,441	28,111	8,304
Nov	3,318	4,689	9,668	8,397	323	18,388	7,408
Dec	4,290	9,381	12,287	10,697	305	23,289	13,609
1977							
Jan	7,404	9,870	15,520	11,418	622	16,560	10,969
Feb	6,032	40,602	12,952	23,988	961	37,901	63,170
Mar	15,158	94,290	13,186	44,055	4,137	61,378	65,402
Apr	11,486	41,094	11,458	46,210	5,040	62,708	42,740
May	5,508	7,987	13,775	11,214	1,533	26,522	9,877
June	4,630	6,466	14,602	9,626	686	24,914	7,925
July	3,314	4,812	10,519	8,103	153	18,775	6,217

*May 16-31

Units are kg as N or P.

(Includes Rockwall sewage treatment plant and atmospheric loadings.)

TABLE 34 ANNUAL TRIBUTARY AND ATMOSPHERIC LOADS*

Station		Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
Forney Pump Station	(13)	554	6,480	2,434	8,865	538	11,837	37,266
Rowlett Creek at Miller Road**	(14)	75,635	162,629	30,216	110,780	12,626	153,622	202,279
Rowlett Creek at Hwy 78	(15)	27,376	136,247	22,270	89,468	9,975	121,713	148,651
Muddy Creek at Liberty Grove Road	(16)	3,586	25,937	592	20,180	3,796	24,568	29,489
Cottonwood Creek at Stonewall Road	(17)	211	1,878	39	1,085	41	1,164	1,794
Rush Creek at Troy Road	(18)	54	1,307	65	1,213	25	1,303	1,576
East Fork of the Trinity River at Hwy 78	(19)	0	3	0	1	0	1	11
Camp Creek at Hwy 205	(20)	516	16,884	470	5,300	346	6,116	15,410
Squabble Creek at Hwy 205	(21)	115	1,646	47	1,138	39	1,224	2,304
Lake Ray Hubbard Spillway	(22)	1,921	25,772	2,465	21,717	2,279	26,463	134,400
Yankee Creek at Meadowview Road	(23)	10	36	7	87	2	96	143
Garland Sewage Treatment Plant	(24)	38,439	45,535	33,329	12,966	1,110	47,406	24,048
Rockwall Sewage Treatment Plant		6,300	8,868	9,480	0	0	9,480	3,156
Atmosphere		864	8,676	78,120	78,120	0	156,240	52,080
Total Load		77,471	247,017	144,419	209,558	15,334	369,311	278,662
Output (13 & 22)		2,475	32,252	4,899	30,582	2,817	38,298	166,666
Input-Output		74,996	214,765	139,520	178,976	12,517	331,013	111,996

*Annual totals calculated from 6/76 through 5/77.
**Not used in total load calculations.
Units are kg as N or P.

TABLE 35. PERCENT CONTRIBUTION OF ANNUAL NUTRIENT LOADS BY EACH TRIBUTARY

Station	Soluble Orthophosphate	Total Phosphate	Ammonia	Nitrate	Nitrite	Inorganic Nitrogen	Organic Nitrogen
Rowlett Creek at Hwy 78 (15)	35.4	55.2	15.5	42.8	65.0	33.0	53.4
Muddy Creek at Liberty Grove Road (16)	4.6	10.5	0.4	9.6	24.7	6.7	10.6
Cottonwood Creek at Stonewall Road (17)	0.3	0.8	0	0.5	0.3	0.3	0.6
Rush Creek at Troy Road (18)	0	0.5	0	0.6	0.2	0.4	0.6
East Fork of the Trinity River at Hwy 78 (19)	0	0	0	0	0	0	0
Camp Creek at Hwy 205 (20)	0.7	6.8	0.3	2.5	2.3	1.7	5.5
Squabble Creek at Hwy 205 (21)	0.1	0.7	0	0.5	0.3	0.3	0.8
Yankee Creek at Meadowview Road (23)	0	0	0	0	0	0	0
Garland Sewage Treatment Plant (24)	49.7	18.4	23.1	6.2	7.2	12.7	8.6
Rockwall Sewage Treatment Plant	8.1	3.6	6.6	0	0	2.6	1.1
Atmosphere	1.1	3.5	54.1	37.3	0	42.3	18.8
Rowlett Creek (15 & 24)	85.1	73.6	38.6	49.0	72.2	45.8	62.0
Percent Nutrient Load Leaving the Lake							
Forney Pump Station (13)	0.7	2.6	1.7	4.2	3.5	3.2	11.6
Lake Ray Hubbard Spillway (22)	2.5	10.4	1.7	10.4	14.9	7.2	48.2
(13 & 22)	3.2	13.0	3.4	14.6	18.4	10.4	59.8

which gives a total annual load to Lake Ray Hubbard of 247,017 kg P. Examination of Table 35 shows that Rowlett Creek (Stations 15 and 24) contributed 85.1 and 73.6 percent of the annual loads of soluble ortho P and total P, respectively. These loads were derived from three sources: the Garland-Rowlett sewage treatment plant (Station 24) which contributed 49.7 and 18.4 percent of the annual loads of soluble ortho P and total P, respectively; the Plano sewage treatment plant; and land runoff to Rowlett Creek above the Highway 78 bridge (Station 15).

Table 36 presents the percent contribution of each tributary to the annual tributary phosphorus load to the lake. This load does not include the atmospheric input to the lake. Examination of this table shows that 76.3 percent of the tributary P load was derived from Rowlett Creek (Stations 15 and 24). The total P load at Station 15 (57.2 percent of the total tributary load) was derived from the Plano sewage treatment plant and land runoff above the Highway 78 bridge. It was estimated that the Plano sewage treatment plant contributed 23.8 percent of the total annual tributary load of phosphorus to Lake Ray Hubbard. This estimate was based on a Plano population of 51,800 and an average of 1.1 kg P/capita/yr in the effluent of the sewage treatment plant. It was estimated that the Plano sewage treatment plant effluent would have a lower per capita phosphorus load than the Rockwall sewage treatment plant effluent so a lower loading coefficient (1.1 P/capita/yr) was used to calculate the P load. This was based on the fact that Plano has recently put into operation a new sewage treatment plant which should, if operated properly, result in a lower phosphorus load compared to the Rockwall plant.

The percent contribution of soluble ortho P from the sewage treatment plant to Lake Ray Hubbard could not be estimated because soluble ortho P is likely converted to other P forms in Rowlett Creek prior to the time it reaches the lake. The nutrient load of Muddy Creek as measured at Liberty Grove Road (Station 16) was derived from land runoff to Muddy Creek and the effluent

TABLE 36. PERCENT CONTRIBUTION OF ANNUAL TRIBUTARY PHOSPHORUS LOADS*

Station		Soluble Orthophosphate	Total Phosphorus
Rowlett Creek at Hwy 78	(15)	35.7	57.2
Muddy Creek at Liberty Grove Road	(16)	4.7	10.9
Cottonwood Creek at Stonewall Road	(17)	0.3	0.8
Rush Creek at Troy Road	(18)	0	0.5
East Fork of the Trinity River at Hwy 78	(19)	0	0
Camp Creek at Hwy 205	(20)	0.7	7.1
Squabble Creek at Hwy 205	(21)	0.2	0.7
Yankee Creek at Meadowview Road	(23)	0	0
Garland Sewage Treatment Plant	(24)	50.2	19.1
Rockwall Sewage Treatment Plant		8.2	3.7
Rowlett Creek (15 & 24)		85.9	76.3
Plano Sewage Treatment Plant (estimate)		-	23.8
Wylie Sewage Treatment Plant (estimate)		-	1.4

*Does not include atmospheric loads.

from the Wylie sewage treatment plant. Using a population of 3100 for Wylie and an average of 1.1 kg P/capita/yr in the sewage treatment plant effluent, it was estimated that the Wylie plant contributed approximately 1.4 percent of the total tributary phosphorus load to the lake. As discussed previously, Squabble Creek receives effluent from the Rockwall sewage treatment plant below the tributary sampling point (Station 21). Based on estimates previously presented, the Rockwall sewage treatment plant contributed approximately 3.7 percent of the total P load to the lake. Using an average of 1.4 kg/capita/yr of soluble ortho P, the Rockwall sewage treatment plant contributed 8.2 percent of the total tributary soluble ortho P load to the lake.

The phosphorus loads from the other tributaries were insignificant compared to the loads from Rowlett, Muddy, and Squabble Creeks. A small package sewage treatment plant discharges into Yankee Creek. However, the phosphorus contribution from the effluent of this plant was less than 0.1 percent of the total phosphorus load to the lake from the tributaries. The other tributaries (Cottonwood, Rush, and Camp Creeks) contributed less than one percent of the total P load to the lake. The East Fork of the Trinity River contributed less than 0.1 percent of the total P load to the lake during the study period since essentially no water entered Lake Ray Hubbard from the East Fork due to the raising of the water level in Lake Lavon. Under normal conditions, the East Fork of the Trinity River would contribute approximately 20,000 kg P/yr to Lake Ray Hubbard (US EPA, 1977a). This would be approximately eight percent of the tributary total P load to the lake.

The basis of the US EPA estimate of the nutrient output from Lake Lavon to Lake Ray Hubbard was not discussed in the US EPA report. It is probably based on measured values during 1974. In order to estimate the more general conditions that might prevail for the transport of nutrients from Lake Lavon to Lake Ray

Hubbard, the Dallas Water Utilities provided an estimate of the average flow from Lake Lavon to Lake Ray Hubbard of 1.9 x 10^{11} l/yr. In 1974, the average total P content of Lake Lavon water was found by the US EPA to be 0.068 mg P/l. This concentration and flow gives an estimated 1.3 x 10^4 kg P/yr as the normal contribution from Lake Lavon to Lake Ray Hubbard. This number is approximately one-half of that estimated by the US EPA for 1974. Since both estimates must be based on the US EPA total P concentration data for Lake Lavon in 1974, the difference between the two estimates must be related to the flow. It is expected that the annual average concentration of total P within Lake Lavon will not change drastically over a period of a few years. However, the year-to-year flow from Lake Lavon to Lake Ray Hubbard will be highly variable. Therefore, the total P load from Lake Lavon to Lake Ray Hubbard will also be highly variable.

A summary of the total phosphorus loadings from the sewage treatment plants which discharge into Lake Ray Hubbard is presented in Table 37. The annual total P loading from all sewage treatment plants was 114,603 kg P, which was 46 percent of the total annual loading of 247,017 kg P. If the total P loading from the sewage treatment plants was reduced by 85 percent by advanced wastewater treatment, the total P loading to the lake would be reduced by 39 percent. A reduction in the sewage treatment plant loads of 35 percent, which would be expected if phosphate detergents were banned in the Lake Ray Hubbard watershed (Lee, 1976), would result in a reduction in the total P load to the lake of approximately 15 percent.

It is of interest to compare the estimates of the nutrient loads to Lake Ray Hubbard based on measurements at Station 14 to those based on measurements made at Stations 15 and 24. As noted in previous discussions, Station 14 is not a suitable place for estimating nutrient transport to the lake because it is located in a backwater area of Lake Ray Hubbard. This results in periodic changes in the flow at this location. Another complicating factor associated with sampling Station 14 is that

TABLE 37. TOTAL PHOSPHORUS LOAD TO LAKE RAY HUBBARD FROM SEWAGE TREATMENT PLANTS

Sewage Treatment Plant	Total P Load (kg P/yr)	Percent of Total Annual Load
Garland	45,535	18.4
Plano	56,700	23.0
Rockwall	8,868	3.6
Wylie	3,500	1.4
TOTAL	114,603	46.4

it was frequently stratified. To sample this station properly, a large number of samples would be required.

Examination of Table 34 shows that the total phosphorus load to Lake Ray Hubbard from Rowlett Creek, based on measurements at Station 14, was 162,629 kg/yr. The corresponding estimate, based on summing Stations 15 and 24, was 181,782 kg/yr, which is approximately 20,000 kg/yr greater than the estimated nutrient load using Station 14 data. These results indicate that the possibility of large amounts of phosphorus entering Rowlett Creek below the Highway 78 bridge, other than from the Garland-Rowlett sewage treatment plant, is small. They also indicate that there is possibly substantial phosphorus removed in Rowlett Creek between Highway 78 and Station 14. This would be expected as a result of incorporation of phosphorus into the sediments. This phosphorus would move to Lake Ray Hubbard as part of the particulate phosphorus or in the bed load. In any event, either of these modes of transport would be difficult to sample properly. Failure to sample the particulate phosphorus properly is of minor consequence to this study since, as discussed in a subsequent section of this report, little of the particulate phosphorus entering the lake would likely become available to stimulate algal growth. For further discussion of transport of particulate phosphorus in lakes, see Lee (in press).

Nitrogen

Inorganic nitrogen, which is the sum of nitrate, nitrite, and ammonia, is readily available for algal uptake. As shown in Table 33, the inorganic nitrogen loads from all tributaries and the atmosphere ranged from 16,600 to 62,700 kg N/month. During the dry season, the monthly loadings were between 16,000 and 29,000 kg N, whereas during February, 38,000 kg N entered Lake Ray Hubbard and during March and April, over 60,000 kg N entered the lake each month. These increased nitrogen values are directly related to transport of N compounds in land runoff. Over 90 percent of the inorganic nitrogen was in the forms of ammonia and nitrate, with nitrite contributing less than 10 percent to the total load. During the dry months, the ammonia load was normally larger than the nitrate load; however, during February, March, and April, the nitrate load was 100 to 400 percent larger than the ammonia load.

The sewage treatment plants contribute a large amount of ammonia which is only slowly oxidized to nitrate under oxic conditions. Since most of the tributary flow was derived from the sewage treatment plants during the dry months, the ammonia concentration exceeded the nitrate concentration in the tributaries. During the wet months, when most of the flow was derived from land runoff, the nitrate concentration exceeded the ammonia concentration. During low flow conditions, dischange of wastewater effluents from Plano, Garland, and Rockwall, even under optimum performance of the treatment plants, would be expected to result in significant water quality degradation in the streams due to residual oxygen demand in the effluent. This is a result of the fact that under low flow conditions, the flow in these tributaries is essentially 100 percent wastewater effluent. The residual BOD in these effluents would, under normal conditions, result in significant oxygen depletion in the streams which would prevent nitrification reactions, allowing the conversion of ammonia to nitrate. This conversion exerts a significant oxygen demand on the streams and would lead to a degraded water quality unless the effluents in the sewage treatment plants were nitrified prior to discharge.

Based on the work of Cowen, Sirisinha, and Lee (1976), it is assumed that approximately 80 percent of the organic nitrogen will be converted to forms available for algal uptake within two months of entering the stream. The organic N load from the tributaries and the atmosphere ranged from 6200 to 65,400 kg N/month. During the low flow periods, 6200 to 22,100 kg N were transported to the lake each month whereas during the high flow periods, 34,600 to 65,400 kg N entered the lake each month. The ratio of inorganic N to organic N ranged from 0.6 to 3.1 indicating that approximately 50 percent of the nitrogen entering the lake was in forms readily available for algal uptake.

As shown in Table 34, the total annual load of inorganic N was 369,311 kg N. Approximately 40 percent of the inorganic N (156,240 kg N/yr) was estimated to be derived from the atmosphere with the remaining 60 percent (213,071 kg N/yr) transported to the lake by the tributaries. The tributaries transported 226,582 kg N/yr, and the atmosphere contributed 52,080 kg N/yr which gives a total annual organic N load of 278,662 kg N. Rowlett Creek contributed approximately 46 percent of the total annual inorganic N and 62 percent of the organic N load as shown in Table 35. The Garland-Rowlett sewage treatment plant contributed approximately 13 percent of the total annual inorganic N load to the lake.

On a total nitrogen load basis, the Garland-Rowlett sewage treatment plant contributed 11 percent of the total annual nitrogen load. Using a per capita contribution of 3.4 kg/yr of total nitrogen, the Plano sewage treatment plant contributed about 27 percent and the Rockwall sewage treatment plant contributed approximately 2 percent of the total annual nitrogen load to Lake Ray Hubbard. Approximately 32 percent of the total nitrogen load was derived from the atmosphere. The results of these computations show that over 70 percent of the total N contributed to Lake Ray Hubbard was derived from sewage treatment plants and the atmosphere.

The fact that over 40 percent of the nitrogen is derived from domestic wastewaters is of limited consequence in terms of controlling excessive fertilization of Lake Ray Hubbard from two points of view. In general, algal growth in the Main Body of the lake was not limited by the available nitrogen content, and it is not likely to become nitrogen limited if nitrogen removal practices are initiated at the Plano and Garland domestic wastewater treatment plants. Nitrogen removal from domestic wastewaters is technically feasible, but it is rarely practiced.

While phosphorus removal is routinely practiced in many municipal wastewater treatment plants in the US, Canada, and Western Europe, there are few, if any, domestic wastewater treatment plants that routinely practice nitrogen removal. Further, the cost of nitrogen removal is considerably greater than the cost of phosphorus removal. Therefore, limiting the nitrogen input to Rowlett Arm could at some times help control excessive fertilization of this area, but it would have little or no impact on excessive fertilization of the Main Body of the lake. From the data available, it appears that initiating nitrogen removal at the domestic wastewater treatment plants of Plano, Garland-Rowlett, and Rockwall would not reduce the total nitrogen in the Main Body to a sufficient degree to cause this part of the lake to become nitrogen limited during the critical period of excessive algal growth in late summer. Therefore, it appears that the only feasible management strategy that should be considered for controlling the excessive fertilization of the Main Body of Lake Ray Hubbard is the control of phosphorus from domestic wastewaters. The control of nitrogen and phosphorus from the atmosphere and land runoff, while technically feasible, is not economically or politically feasible at this time.

Groundwater - Septic Tanks

The estimates of nutrient loads for Lake Ray Hubbard do not include estimates of nitrogen and phosphorus loads from groundwater. Many water bodies receive significant amounts of nitrogen but rarely do water bodies receive significant inputs of phosphate from groundwaters. Jones and Lee (1977) have recently completed a comprehensive review on the transport of phosphorus in groundwater systems, with particular reference to the significance of septic tank wastewater disposal systems as a source of phosphorus for lakes. They have concluded that, in general, the amount of phosphorus that a lake would receive from groundwaters, including septic tank wastewater disposal systems, would usually be very low. This is especially true in areas which have calcareous soils or soils with high clay content such as those found in the Lake Ray Hubbard watershed. It is therefore concluded that the septic tank wastewater disposal systems located in the watershed are likely contributing a relatively small amount of phosphorus to Lake Ray Hubbard.

Some of the soils in the watershed are not suitable for septic tank wastewater disposal systems because of their highly impervious nature. Installation of septic tanks in these areas usually results in some effluent coming to the surface of the ground and flowing over the surface toward the nearby watercourse. Normally, however, a combination of seepage into the ground, evaporation, and evapotranspiration will prevent the effluent from moving large distances over the surface. The phosphate present in the effluent that reached the surface would likely be fixed by the soils and taken up by the plants. During periods of heavy rain with overland flow, some of the phosphorus would be carried to a nearby stream in the particulate form or in the form of plant debris (detritus). Based on the work of Cowen and Lee (1976a) most of this would likely not become available for algal growth in the lake. Therefore, it is concluded that septic tank systems which are not functioning properly are not likely to be a significant source of available phosphorus for Lake Ray Hubbard.

Lake Ray Hubbard algal growth appears to be limited by phosphorus except for Rowlett Arm. This situation lessens the significance of any nitrogen transported to the lake via groundwaters. While the groundwater table in a large part of the watershed is relatively shallow (10 to 30 feet deep), it does not appear that significant amounts of groundwater are contributed to the tributaries of Lake Ray Hubbard during low flow since many of these tributaries dry up completely. Forrest and Cotton Engineers (1960) concluded that there would likely be limited groundwater input to the lake. This seems to be a reasonable assessment of this situation. It is therefore concluded that groundwaters and septic tank wastewater disposal systems are likely contributing very small amounts of nitrogen to Lake Ray Hubbard. It should be noted that this conclusion may not be applicable to other water bodies in the Dallas-Fort Worth region. For any water body for which there is appreciable groundwater input, this input should be considered as a potential significant source of nitrate. As discussed by Sonzogni and Lee (1972), it is important to determine how the groundwater enters the lake in order to assess whether any nitrate present in it would actually be available to stimulate algal growth. Particular emphasis must be placed on evaluating the interaction between the nitrate in the groundwater and anoxic lake sediments. Under these conditions, nitrate can be converted to nitrogen gas through denitrification reactions.

Nutrient Export Coefficients

Nutrient export coefficients are measures of the amounts of nutrients (N and P) derived from the land. If the nutrient export coefficients are known for the different types of land use in a watershed the N and P load to a lake can be estimated without actually measuring the flows and concentrations of N and P in the tributaries to the lake. The nutrient export coefficients were calculated for the watersheds of the tributaries discharging into Lake Ray Hubbard by dividing the total P, soluble ortho P and total N (inorganic N + organic N) annual loads from land runoff by the area of the watersheds. The N and P contributions from the Plano and Garland sewage treatment plants were subtracted from the Rowlett Creek load and the Wylie sewage treatment plant load was subtracted from the Muddy Creek load since the nutrient export coefficients are measures of the amounts of nutrients derived from land runoff. There were insufficient data to calculate nutrient export coefficients for the various types of land use in the watersheds. Originally, it was thought that some estimates of the effect of land use on nutrient export coefficients could be obtained by comparison of the nutrient export coefficients obtained for each of the tributaries, making use of the fact that some of the tributaries have a greater percentage of urban area than others. Examination of Table 1 shows that except for Camp Creek, about 8 to 18 percent of the land in the watershed's tributaries is classified as urban. From 0.5 to about 5% of the land use is devoted to forest, with the remainder devoted to rural land in crops or grass. There are insufficient differences in land use for the various tributaries to begin to assign expected nutrient export coefficients to various types of land use.

Table 38 presents the values obtained in this study and the values obtained from the National Eutrophication Survey studies on Lake Lavon and Lake Lewisville (US EPA, 1977a,b). The export coefficients derived by Sonzogni and Lee (1972) and Rast and Lee (1977) are presented for comparison with the Lake Ray Hubbard values. The total P export coefficient ranged from 8 kg P/km^2/yr for the Yankee Creek watershed to 255 kg P/km^2/yr for the Rowlett Creek watershed. As shown in the table, the values obtained in the Lake Ray Hubbard watershed are considerably higher than the NES values obtained for Lake Lavon and Lake Lewisville (7 to 54 kg P/km^2/yr). Sonzogni and Lee (1972) found a total P export coefficient of 65 kg P/km^2/yr from rural land for the watersheds of the Madison, Wisconsin lakes. Rast and Lee (1977) reported values ranging from 30 to 70 kg P/km^2/yr from rural land in the watersheds of the US OECD eutrophication study lakes and other US water bodies.

Prior to the Lake Ray Hubbard study period there had been several years of rainfall which resulted in limited runoff from some parts of the watershed. Some tanks (farm ponds) on the tributaries did not completely fill during this period so the phosphorus entering these tanks was retained in the tanks until the spring of 1977. This situation is discussed further in the Discussion section of this report. During the spring of 1977 there was sufficient rainfall to fill the tanks, and water and associated nutrients from some parts of the watershed entered tributaries of Lake Ray Hubbard for the first time in almost two years. As discussed previously, the phosphorus load was large

TABLE 38 . NUTRIENT EXPORT COEFFICIENTS

Tributary	Total P Export Coef. (kg P/km^2/yr)	Soluble Ortho P (kg P/km^2/yr)	Total N Export Coef. (kg N/km^2/yr)
Rowlett Creek	255	42	415
Muddy Creek	218	8	443
Cottonwood Creek	217	24	342
Rush Creek	109	5	240
Camp Creek	220	7	280
Squabble Creek	114	8	245
Yankee Creek	8	2.2	53
US EPA (1977a,b)			
Lake Lavon	7-54		272-576
Lake Lewisville	7-45		82-360
Sonzogni & Lee (1972)	65	32	-
Rast & Lee (1977)	30-70		500-1000

Dash (-) indicates not determined.

during the spring months with over 80 percent of the total annual P load entering the lake during February, March,and April. This was likely due to the release of water from the tanks and to runoff from the land during periods of heavy rainfall. This situation can possibly explain why the total P export coefficients obtained in this study were higher than what would be expected during normal years.

It should be noted that frequently year-to-year variations, especially for total phosphorus, can be quite large. As discussed by Sonzogni and Lee (1972), the amount of total phosphorus derived from the land is highly dependent on rainfall frequency, duration, and intensity as well as other factors. There is no reason to believe that total P contributions from this region should, on the average, be markedly different from what is found in many other parts of the US. This is supported by the fact that the 1974 US EPA (US EPA, 1977a,b) studies of several watersheds of this region showed that the total P nutrient export coefficients were typical of what were found in many parts of the US. It is therefore concluded that the abnormally high total phosphorus export coefficients found in the Lake Ray Hubbard watershed during 1976-77 (principally spring 1977) are most likely due to the somewhat atypical climate and hydrology during the past several years.

The soluble ortho P nutrient export coefficients ranged from 2.2 kg P/km^2/yr in the Yankee Creek watershed to 42 kg P/km^2/yr in the Rowlett Creek watershed. These values are in the range of the export coefficient of 32 kg P/km^2/yr reported by Sonzogni and Lee (1972) in the watersheds of the Madison lakes. It therefore appears tnat the primary discrepancy between the results of this study and the US EPA studies on Lakes Lavon and Lewisville and studies in other parts of the US is with the total phosphorus values.

The total N export coefficients ranged from 53 kg N/km^2/yr in the Yankee Creek watershed to 443 kg N/km^2/yr in the Muddy Creek watershed. These values are comparable to the values reported by the US EPA (1977a,b) in the NES reports on Lake Lavon (272 to 576 kg N/km^2/yr) and Lake Lewisville (82 to 360 kg N/km^2/yr). Rast and Lee (1977) reported a range from 500 to 1000 kg N/km^2/yr for the export coefficients of the watersheds of the US OECD water bodies. The Lake Ray Hubbard nitrogen export coefficients were not significantly higher than the values reported for other water bodies by other investigators. Ammonia is readily oxidized to nitrate which is transported via groundwater to the soil and is therefore not retained at the surface to the same extent as phosphate. Since the nitrate is not retained at the surface, it does not run off to the same degree as phosphate when a heavy rainfall occurs. Therefore the total nitrogen coefficients during the study period are typical and were not affected to the same degree as the P export coefficients by the previous years' climate.

Since the total phosphorus export coefficients calculated for the watersheds of the tributaries of Lake Ray Hubbard appear to be abnormally high for the study period, they should not be used for estimating nutrient loads to other water bodies in the north central Texas region. The nitrogen export coefficients found during the study are typical of those found for other lakes in this region and in other areas of the country so an average of these values should be used for estimating the nitrogen loads to other lakes in the area.

It is recommended that the following nutrient export coefficients be used for water bodies in the north central Texas region - 50 kg total P/km^2/yr and 350 kg total N/km^2/yr. It is evident that additional work needs to be done to determine nutrient export coefficients from various types of land use in this region. Also, work needs to be done on determining the amounts of total N and P present in streams of this region which would become available to support algal growth.

SECTION 7

MODELING

MODELING APPROACH*

Introduction

One of the goals of the Lake Ray Hubbard study is the development of a eutrophication model that will describe the relationship between nutrient loads and lake response in water bodies located in the north central Texas region near Dallas. The primary purpose of the modeling effort was to provide a basis for estimating the potential impact of altering the nutrient load to Lake Ray Hubbard on the water quality within the lake. It was thought that this model would be applicable to water bodies located throughout the southern US. After verification, this model could then be used as a tool in water quality management. After reviewing the various eutrophication models currently available, it was decided that the best approach to take was that of using the basic structure of the Lake I model developed under a grant from the US EPA by Thomann, DiToro, O'Connor and others at Manhattan College (Thomann et al., 1975). This model attempts to formulate, using a series of differential equations and a mass balance approach, the major nutrient fluxes and the physical, chemical, and biological processes that influence the amount of chlorophyll present in the water.

In the Lake I model, the modelers' intent was "...to structure a mathematical model framework of the major features of eutrophication in large lakes." It should be noted that their purpose was investigation of large lakes and that data for model parameterization and verification was collected on Lake Ontario, a north temperate lake. The model embodies chemical and biological interactions important in lake eutrophication and has proven useful in estimating response behavior (expressed as chlorophyll) under known nutrient loadings for the lakes upon which it has been tested.

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*This section was coauthored by Magda Abdul-Rahman and Dr. William Cale.

Lake I considers the different species of nitrogen and phosphorus as potential limiting factors. Ambient temperature, available solar energy, and nutrients determine phytoplankton growth rate. Herbivorous and carnivorous zooplankton are modeled explicitly to complete a simple food chain. The model lake is vertically stratified into three segments with appropriate parameters for sinking and upwelling of nutrients and biota. Knowledge of loading dynamics and lake morphometry and hydrology is required to run the model.

The Lake I model requires a large amount of data in the form of inputs, initial conditions, and rate constants for solving the numerous differential equations that make up the model. The Manhattan College group followed the approach of selecting rate constants and other needed information within the range of values typically found in laboratory and other studies. Often there is a considerable range of values that can be utilized; one of the first steps in a modeling effort of this type is to adjust the magnitude of the various constants (tuning the model) so that input of measured nutrient loads results in accurate prediction of measured chlorophyll and nutrient concentrations in the lake. Because the climatic regime in Texas is markedly different from that of the north temperate US, modifications of some of these constants have been made in an attempt to get a proper input-output simulation for Lake Ray Hubbard.

The discussion of the modeling approach and results presented in this section of the report represent the first phase of an ongoing study that is being conducted by the authors. Progress within the initial phase has been significantly hampered by the lack of reliable flow data which could be used as a basis for estimating nutrient loads and the very short time between the completion of sample collection and analysis (about September 1, 1977) and the date at which the draft report had to be submitted to the Dallas Water Utilities. It is contemplated that considerable additional work will be done on modeling nutrient load response relationships for Lake Ray Hubbard after submission of this report. Those who review this section and wish to have additional information on the subsequent work should contact the authors. It is anticipated that the additional studies will be presented in a Ph.D. dissertation by Magda Abdul-Rahman.

Modeling Philosophy

The mathematical framework for this type of model is derived from empirical and theoretical relationships developed both in the field and in the laboratory. These relationships describe the dynamics of phytoplankton communities based on environmental parameters such as light, temperature, nutrient availability, and grazing by higher trophic levels. The dynamics of zooplankton kinetics are described as a function of temperature, food density, and predation. These relationships, although subject to continual refinement, appear to have a sound basis and are general enough to cover many situations.

Development of the Modeling Framework

The principle of conservation of mass forms the basis for formulation of the equations which describe constituents of importance in the eutrophication process. The essential element of this process is the dynamic behavior of the phytoplankton. In their simplest form these equations relate the growth of phytoplankton to the availability of nutrients; in their complex form they may incorporate interaction among many nutrients and predator-prey relationships between contiguous trophic levels. The nutrients in various forms are supplied by inputs from man-made and natural sources, and also by recycling from internal sources due to death and decay of organisms in the system. The differential equations are developed by applying the principle of conservation of mass, from which quantitative relationships are developed relating the progressive changes of state in the various systems. The basic systems are nutrients, phytoplankton, and zooplankton. The mathematical expression of conservation of mass, which includes terms to describe both mass transport phenomena and source terms, is a differential equation of the following form:

$$\frac{dc}{dt} = \nabla \cdot E \nabla C - \nabla \cdot QC + Sc + \frac{Wc}{V}$$

where: C = concentration of the substance of interest
E = diagonal matrix of dispersion coefficients
Q = advective flow rate vector
∇ = gradient operator
(Giving the partial differential of any parameter in the three dimensions

$$E = \frac{\partial E}{\partial x} + \frac{\partial E}{\partial y} + \frac{\partial E}{\partial Z})$$

Sc = vector whose terms are the rate of mass addition by the sources and the sinks
Wc = direct inputs of the substance C
V = volume of water body

This equation states that the concentration of a substance in a water body changes as a result of 1) dispersion caused by wind action, turbulent diffusion, and density differences in the water column; 2) advection, assuming a unidirectional flow, Q, accounting for differences between input and output concentrations; 3) the rate of mass addition by the sources and sinks within the lake due to biological and chemical interactions; and 4) the external load of substance, C, to the water body.

Because of the complexity of the interactive system and the computation time involved in obtaining solutions in the three dimensional system, Lake I assumes that Lake Ontario is well mixed horizontally, and gradients are allowed to develop only in the vertical dimension. The last equation is then simplified to include:

a) horizontal transport,
b) vertical dispersion between epilimnion and hypolimnion,
c) no mixing between the bottom layer and hypolimnion so that the bottom layer-sediments represent a sink to the system.

In the Lake I model it is assumed that the lake is well mixed in the winter and spring, stratified during the summer, and becomes well mixed again during the fall overturn. These conditions are simulated by control of the vertical mixing term.

The general equation is then reduced to:

$$\frac{dC1}{dt} = \frac{(Qi \cdot C01 - Qo \cdot C1)}{V} - \frac{EA \cdot AR}{V \cdot \Delta Z}(C12 - C1) + SC1 + \frac{Wc}{V}$$

where:
Qi = input flow
Qo = output flow
EA = dispersion coefficient
AR = area through which the dispersion takes place
ΔZ = depth of segment
$C01$ = boundary concentration
$C12$ = concentration in the lower layer
$C1$ = equilibrium concentration in the upper layer
$SC1$ = source sink kinetic term
Wc = external input of C1

Thus, the change in concentration of a substance, C1, is equal to the boundary concentration entering the water body minus the equilibrium concentration going out minus what is lost to the lower layer through molecular dispersion, plus what is gained through biological and chemical reactions (kinetic term), plus an external input (Wc) of C1 to the water body.

For the Lake Ray Hubbard simulation, it was assumed that the Main Body of the lake is well mixed horizontally; vertical stratification is allowed during the summer at a depth of 10 meters. Dispersion takes place between a thin benthos layer and the water column during the whole year. During the summer, the thermocline prevents the transport of released nutrients from the benthos layer to the surface waters. As indicated above, the Lake I model does not make provisions for the return of nutrients from the sediments. This may be appropriate for a simplified model of Lake Ontario, which has a depth of 90 m, but it is likely to be inappropriate for Lake Ray Hubbard which is much shallower. As discussed by Lee (1970) a shallow water column promotes release of nutrients through wind and organism induced mixing of the sediments. The differential equation describing the change in concentration of a substance is solved for each system given the initial conditions. The source-sink term, Sc, is defined and formulated separately for each particular system.

Phytoplankton Chlorophyll System

The basic kinetic interaction for phytoplankton chlorophyll is given by:

$$Sp = V\,(GP - DP)\cdot P$$

where: P = phytoplankton concentration
Sp = kinetic term (net change)
GP = growth rate of phytoplankton
DP = death rate of phytoplankton

The growth rate of phytoplankton in natural waters is a complicated function of the species of phytoplankton present and the different reactions to solar radiation and temperature, and the balance between nutrient availability and phytoplankton requirements. In order to derive a growth rate function the population is characterized as a whole. The biomass of phytoplankton is expressed in terms of chlorophyll a concentration. The growth rate expression is conceptually given by:

$$GP = \underbrace{G1\ (Gmax,T)}_{\text{Temperature Influence}} \cdot \underbrace{[I(Ia,Is,SKE,P,H)]}_{\text{Light Influence}} \cdot \underbrace{[N'(Kmn,N2,N3)][Kmp,P2])}_{\text{Nutrient Influence}}$$

where: GP = realized growth rate
Gmax= max growth rate
T = water temperature(°C)
Is = light intensity at G max
Ia = incoming surface solar radiation
SKE = extinction coefficient
H = depth of water column
Kmn = half saturation constant for total inorganic nitrogen N'
N2 = ammonia nitrogen concentration
N3 = nitrate nitrogen concentration
Kmp = half saturation constant for phosphorus
P2 = available phosphorus

Temperature Effect--

The temperature effect is given by Eppley (1972)

$$G1(T) = Gmax\ (1.066)^{T}$$

where Eppley reports G max equal to 0.587 (day^{-1})

Light Effect--

The light intensity to which the phytoplankton are exposed is not uniformly at the optimum value but varies as a function of depth due to natural turbidity in the water column and as a function of time over the day. Thus the phytoplankton in the lower layers are exposed to intensities below optimum, and those at the surface may be exposed to intensities above the optimum so their growth rates would be inhibited. In order to calculate the light intensity, at a certain depth, Z, in the water column, V, the extinction coefficient must be determined. The intensity at depth Z, I(Z), is related to the surface intensity, Io, by the formula:

$$I(Z) = Io \cdot e^{-SKE \cdot Z} \tag{1}$$

The growth rate of phytoplankton as a function of light intensity is given by:

$$F(I) = \frac{I(Z)}{Is} \exp\left(\frac{-I(Z)}{Is} + 1\right) \tag{2}$$

where Is is the saturating light intensity. Combining both effects (Equations 1 and 2) produces the following equation:

$$F(I) = \frac{Io \cdot e^{-SKE \cdot Z}}{Is} \exp\left[-\frac{Io}{Is} e^{-SKE \cdot Z} + 1\right] \tag{3}$$

To apply Equation 3 to finite volume elements, within which the phytoplankton concentration is uniform, it is necessary to average this reduction factor throughout the depth Z. Let Zo and Z1 be the depth at the surface and bottom, respectively, with Zo = 0, Z1 = Z. Equation 3 must then be integrated a second time over a time period f (the photoperiod) so that:

$$Io(t) = Ia \qquad o<t<f$$
$$Io(t) = 0 \qquad f<t<1$$

This assumes an average surface light intensity of Ia during the photoperiod and a zero value otherwise. Letting rj be the reduction in growth rate attributed to nonoptimal light conditions in a volume V averaged over depth and time, Equation 3 is rewritten as:

$$rj = \frac{1}{Z1} \int_{o}^{Z1} \frac{1}{T} \int_{o}^{f} \frac{Io}{Is} \cdot e^{-SKE \cdot Z} \cdot \exp[- \frac{Ia}{Is} e^{-SKE \cdot Z} + 1] \, dt \cdot dZ \tag{4}$$

The solution of the integration of Equation 4 gives:

$$rj = \frac{2.718}{SKE \cdot Z} f \, [e^{- \frac{Ia}{Is} e^{-SKE \cdot Z}} - e^{- \frac{Ia}{Is}}] \tag{5}$$

Equation 5 is the functional form for the light effect on phytoplankton growth used in the Lake I program.

The extinction coefficient is a function of both turbidity and phytoplankton concentration. Thus, an important feedback mechanism exists that inhibits phytoplankton growth. As the concentration of phytoplankton in a volume element increases, the extinction coefficient, particularly in the green wavelengths, starts to increase. This mechanism is called self-shading. A regression was done on Lake Ray Hubbard data, and the following correlation was obtained for the Main Body:

$$SKE = SKEP + 0.0166 \times \text{Chlorophyll concentration}$$

where SKEP is the extinction coefficient attributable to other causes than chlorophyll density.

Nutrients Effect--

The nutrient effect uses the product of Michaelis-Menton kinetics and is given by:

$$N'p = \frac{N2 + N3}{KMN + (N2 + N3)} \cdot \frac{P2}{KMP + P2}$$

The Lake I model assumes that phytoplankton population dynamics follow Monod growth kinetics with respect to the important nutrients. That is, given an adequate level of substrate concentration, the growth rate proceeds at the saturated rate for the temperature and light conditions present. However, at low substrate concentrations, the growth rate decreases in proportion to substrate concentration. Thus, for a nutrient, Ni, the factor by which the saturated growth rate is reduced is $\frac{Nj}{KM + Nj}$. The constant KM (called Michaelis or half saturation constant) is the nutrient concentration at which the growth rate is half the maximum growth rate.

The values of KMN and KMP were chosen from the literature (DiToro *et al.*, 1971). At the present time, silica limitation is not used in this model. The complete growth rate expression is then given by the product of temperature, light and nutrient effect expressions.

Phytoplankton Death Rate--

The factors which affect the death rate of phytoplankton are endogenous respiration, grazing by herbivorous zooplankton, and sinking.

Endogenous Respiration--The endogenous respiration rate of phytoplankton is the rate at which the phytoplankton oxidize their organic carbon to carbon dioxide. There is a net loss of phytoplankton carbon and the population biomass is reduced in size. The respiration rate as a function of temperature has been investigated by the Manhattan College group (Thomann *et al.*, 1975) and reported to be:

$$\text{Death rate due to endogenous respiration} = K2C + K2T^{(T-20^\circ)}$$

The exponent for the Lake Ray Hubbard data was changed to (T-30°) since Lake Ray Hubbard exhibits warmer temperatures than Lake Ontario.

Grazing by Zooplankton--The interaction between phytoplankton and herbivorous zooplankton is given as:

Death due to zooplankton grazing = Cg x Z

where: Cg = grazing rate of herbivorous zooplankton
Z = concentration of herbivorous zooplankton in mg C/l

Phytoplankton Sinking--A significant fraction of phytoplankton population dynamics can be attributed to sinking. Published values of the sinking velocity of phytoplankton, obtained mostly in quiescent laboratory conditions, range from 0.07 to 18 meters/day. In the initial Lake I model of Lake Ontario, the vertical settling velocity was treated as a constant. In Lake Ray Hubbard, simulations of the sinking velocity were altered seasonally between a maximum of 0.2 m in the summer and a minimum of 0.02 m in winter. The sensitivity of the results to the phytoplankton settling velocity is discussed in the model verification section.

The death rate of phytoplankton is then equal to:

$$Dp = K2C + (1.08)^{(T-30^\circ)} CG \cdot GZ + \frac{\text{Sinking Vel.}}{\text{Depth}} \cdot K4$$

where: K4 = sinking rate (day^{-1})

Zooplankton System

This system is analogous to the phytoplankton system. Herbivorous zooplankton grow in accordance with the availability of their food, the phytoplankton. They are preyed upon by the upper levels of the food chain and undergo endogenous respiration and death. Their excretion products may be significant sources of nutrients for the phytoplankton. To account for the fact that zooplankton graze more than they assimilate, a conversion efficiency coefficient is introduced into the equations. In order to simplify the analysis, the predation effect of the upper levels is usually introduced as an empirical constant. The kinetic equation is:

$$SZ = V \cdot (GZ - DZ) \cdot Z$$

$$GZ = AZP \cdot CGZP \cdot KMPL \cdot \frac{P}{KMPL + P}$$

where: AZP = zooplankton carbon to phytoplankton chlorophyll efficiency

CGZP = grazing rate of herbivorous zooplankton

KMPL = half-saturation concentration of phytoplankton at which zooplankton growth is half maximum

This latter term reflects the saturation of zooplankton growth rate by high phytoplankton populations. In the quantity CGZP, P is the maximum rate of phytoplankton biomass ingestion. To convert this value to zooplankton carbon, the above quantity is multiplied by the coefficient AZP which converts phytoplankton biomass ingested to zooplankton biomass produced. The realized growth rate is then determined by multiplying by $\frac{KMPL}{KMPL + P}$.

The mortality of the herbivorous zooplankton is given by two terms: endogenous respiration and grazing by the next higher trophic level.

Therefore,

$$\overset{(1)}{DZ} = \overset{(1)}{KZ}\,(T) + CG21\,(T)\,\overset{(2)}{Z}$$

where: $\overset{(1)}{KZ}$ = endogenous respiration rate as a function of temperature (1)

CG21 = grazing rate (liters/day/mg C C°)

$\overset{(2)}{Z}$ = next highest trophic level designated by superscript (2)

Expressions for the carnivorous zooplankton are similar to the above equations.

$$\overset{(2)}{SZ} = V(\overset{(2)}{GZ} - \overset{(2)}{DZ}) \cdot \overset{(2)}{Z}$$

$$\overset{(2)}{GZ} = AZ \cdot CG21 \cdot \overset{(1)}{Z}$$

$$\overset{(2)}{DZ} = \overset{(2)}{KZ}\ (T) + CG21\ (T)\ \overset{(3)}{Z}$$

Nutrient System

The source term S_{nutrient} in the conservation of mass equation for the concentration of nutrient N is the sum of all sources and sinks of the nutrient within the volume V. The primary interaction between the nutrient system and the phytoplankton system is the reduction of nutrients associated with phytoplankton growth rate GP x P. If ANP is the nutrient-to-phytoplankton biomass ratio of the population, then the loss of the nutrient due to phytoplankton growth is ANP x GP x P.

A secondary interaction between the biological systems and the nutrient systems is excretion of nutrients by zooplankton and release from dead phytoplankton and zooplankton. Using the formulation for zooplankton growth rate, the rate of nutrient excretion is determined from the amounts of nutrients grazed but not assimilated. Thus:

$$\Delta N = ANP \cdot CG \cdot Z \cdot P - ANP \cdot GZ \cdot Z$$

or

$$\Delta N = ANP \cdot Z\ (CG \cdot P - GZ)$$

Substituting for the growth rate of zooplankton (GZ) we have:

$$\Delta N = ANP \cdot Z\ (CG \cdot P - AZP \cdot CG\ \frac{KMPL \cdot P}{KMPL + P})$$

$$\Delta N = ANP \cdot CG \cdot Z \cdot P\ (1 - AZP\ \frac{KMPL}{KMPL + P})$$

where: AZP = carbon to chlorophyll ratio
ANP = nutrient to chlorophyll ratio

At high phytoplankton concentrations, almost all grazed phytoplankton is excreted and the bracketed term in the above equation approaches unity. A similar term is used to determine nutrient release by the death of phytoplankton, ANP x K2 x T X P and that released by the death rate of carnivorous zooplankton, ANZ x K3 x T x A, where ANZ is the nutrient to zooplankton biomass ratio. The organic form is then converted to the inorganic forms.

The Organic Nitrogen System--The major components of the nitrogen system are detrital organic nitrogen, ammonia nitrogen, and nitrate nitrogen. In natural waters there is a stepwise transformation of organic nitrogen to ammonia, nitrite, and nitrate, yielding nutrients for phytoplankton growth. The kinetics of these transformations are assumed to be first order reactions with temperature dependent rate coefficients. The hydrolysis of organic nitrogen produces ammonia, which is then oxidized to nitrite and nitrate.

The source term of the organic nitrogen system is given by:

$$SN1 = ANP \cdot CG \cdot Z \cdot P \left(1 - \frac{AZP \cdot KMPL}{KMPL + P}\right) + ANP\ K2(T) \cdot P$$
$$+ANZ\ K3(T)\ Z - K34(T) \cdot N1 - K33 \cdot N1 \cdot K4$$

where: N1 = organic nitrogen concentration
K34(T) = K34C + K34T * T
is the rate of hydrolysis of organic nitrogen to ammonia(found to be linearly dependent on Temp(1)
K2(T) = K2C + K2T·T
rate of endogenous respiration of phytoplankton (also assumed to be linearly dependent on Temp)
K3(T) = K3C + K3T * T
rate of endogenous respiration of zooplankton
K4 = sinking rate (/day)
K33 = sinking coefficient

The first term in the above equation represents the rate of generation of organic nitrogen from excretion of grazed, but not assimilated, phytoplankton; the second and third terms also represent sources of organic nitrogen from dead algae and zooplankton. The last two terms represent the loss of organic nitrogen due to hydrolysis and the loss due to sinking.

The Ammonia System:--For ammonia the reaction term is given by;

$$SN2 = K34(T) \cdot N1 - K45(T) \cdot N2 - ANP \cdot \alpha \cdot GP \cdot P$$

where: $N2$ = ammonia nitrogen concentration

$K45(T) = K45C + K45T * T$

is the rate of oxidation of ammonia to nitrate and nitrite assumed to be linearly dependent on temperature.

$\alpha = \frac{N2}{N2 + N3}$, a preference factor

The first term represents the ammonia produced from organic nitrogen hydrolysis; the second term represents the loss of ammonia through oxidation to nitrate; the last term represents the amount of ammonia nitrogen removed by the phytoplankton.

The Nitrate System--The reaction term for the nitrate system is:

$$SN3 = K45(T) \cdot N2 - ANP\ (1 - \alpha) \cdot GP \cdot P$$

Thus, nitrate in the system equals the amount generated through ammonia oxidation minus what is consumed by the growing phytoplankton.

The Particulate Phosphorus System--Particulate phosphorus is generated by the death of phytoplankton and zooplankton. The phosphorus in this form is converted to the inorganic state and becomes available to the algae. A sink of particulate phosphorus, like nitrogen, results from settling. The particulate P1 system kinetic term is given by:

$$SP1 = APP \cdot K2(T) \cdot P + APP \cdot AZP \cdot K3(T) \cdot Z - K67(T) \cdot P1$$

$$+ APP \cdot CG \cdot ZG \cdot P \left(1 - \frac{AZP \cdot KMPL}{KMPL + P}\right) - K66 * P1 \cdot K4$$

where APP = phosphorus to chlorophyll ratio
K66 = sinking coefficient
K67 = K67C + K67T * T
rate of oxidation of particulate phosphorus to soluble phosphate (also assumed to be linearly dependent on temperature)

In the above equation, particulate phosphorus is generated by the death of phytoplankton and zooplankton (first two terms) and lost through conversion to inorganic phosphorus (third term). The fourth term represents the gain of particulate phosphorus through excretion of grazed, but not assimilated, phytoplankton. The last term represents the sinking organic particulate matter.

The Inorganic Phosphorus System--It is assumed that inorganic phosphorus is the form available to phytoplankton. The source-sink equation for inorganic phosphorus P2 is given by:

$$SP2 = K67(T) * P1 - APP \cdot GP \cdot P - K77 \cdot P2 \cdot K4$$

where: K67 = rate of decomposition of organic phosphorus to forms available for phytoplankton utilization
K77 = represents an overall loss of inorganic phosphorus usually associated with settling.

The first term represents the rate of generation of orthophosphate from particulate phosphorus. The second term represents the loss due to phytoplankton consumption. The last term is the loss associated with settling.

Model Equations

The differential equations describing the dynamics and chemical variables in each of the eight systems of the model are presented in Table 39.

LAKE I MODEL INPUTS

The simplified model designated Lake I assumes that Lake Ray Hubbard is completely mixed during fall, winter, and spring and stratified from May to August.

Surface Area and Volume of Lake Ray Hubbard

The surface area and volume of Lake Ray Hubbard were calculated for eight pool elevations using a DWU topographic map of the lake basin with contour intervals of 2 meters. The contour map was overlain with a grid which had squares with areas equivalent to 9×10^4 m^2. The surface area was determined by counting the number of squares between each contour line beginning at the 120 m line and ending at the boundary of the lake at the normal pool elevation of 132.4 m. Table 40 presents the surface areas and percent of the total surface area at each pool elevation for the Main Body of Lake Ray Hubbard. The volume at each pool elevation was calculated using the surface area and depth at each elevation. The lake was horizontally divided into seven sections with depths of 2 m. The mean surface area of each section was calculated by determining the mean of the surface areas of the upper and lower contour lines. The mean surface area was multiplied by the average depth to determine the volume of the section. The volumes of the sections were summed to obtain the volume at each pool elevation.

Table 41 gives volumes and the percent of total volume above each pool elevation in the Main Body. Figure 69 shows the hypsometric curves of the Main Body surface area and volume with depth.

TABLE 39. MODEL EQUATIONS

```
(1) * The Phytoplankton Chlorophyll System
    SKE=SKEP+0.0166*C1
    IAV=IAVI/F
    SINK=(SVEL/DEPTH)*K4
    SNK=SINK*C1*VOL
    TEMP1=SKE*DEPTH
    TEMP2=IAV/IS
    TEMP3=EXP*-TEMP1)
    TEMP4=(2.718*F/TEMP1)*(EXP(-TEMP2*TEMP3)-EXP(-TEMP2))
    TEM33=K1C+K1T*STP+(.58*(1.066**STP))
    CN=C4+C5
    XEMP1=CN/(CN+KMN)
    GIT=TEM33*TEMP4
    XEMP2=C7/(C7+KMP)
    GP=(GIT)*(XEMP1*XEMP2)
    GPC1=GP*C1
    * Death Rate
    TEM11=(K2C*(K2T**(STP-20.)))
    TEM22=(CGC+CGT*STP)*C2
    DP+TEM11+TEM22+SINK
    TEMP5=GP-DP
    PHYTOC=C1*CCHL
    C11=(GP-DP)*C1
    *
```

Where: SKE = extinction coefficient

SKEP = extinction coefficient at zero chlorophyll

C1 = chlorophyll a biomass in μg/l

K4 = sink coefficient = 1 (value after Lake Ontario)

IAV = light intensity (ly/day)

(continued)

C4 = concentration of ammonia nitrogen in mg/l

C5 = concentration of nitrate nitrogen in mg/l

GP = growth rate of phytoplankton chlorophyll (/day)

GPC1 = growth of phytoplankton (μg/l/day)

TEMP4 = reduction factor associated with non-optimum light on phytoplankton growth

TEM11 = endogenous respiration rate

TEM22 = death rate due to zooplankton grazing

TEM33 = growth rate as function of temperature

XEMP1 = Michaelis Menton product term for available nitrogen

XEMP2 = Michaelis Menton product term for available phosphorus

DP = death rate of phytoplankton (/day)

SINK = sinking rate of phytoplankton (/day)

```
(2) * System 2 Zooplankton Carbon No. 1
    GZ=AZP*CCHL*TEM22*(C1/C2)*(KMPL/(KMPL+C1))
    * Death Rate
    DZ=K5+K4C+K4T*STP+(CGC8+CGT8+STP)*C8
    C22=(GZ-DZ)*C2
```

Where: GZ = herbivores zooplankton growth rate (/day)

DZ = death rate of herbivores zooplankton (/day)

C2 = concentration of zooplankton carbon (mg/l)

C22 = kinetic concentration of zooplankton in mg C/l)

```
(3) The Organic Nitrogen System
    * Death Rate of Zooplankton
    SR31=NCHL/CCHL*((K8+K8T*STP)*C8+(K5+K4C+K4T*STP)*C2)
    * Grazed but not Assimilated
    SR34=(NCHL/CCHL)*((1.-A8Z)*(CGC8+CGT8*STP)*C2*C8)
    * Grazed but not Assimilated Phytoplankton
    SR32=(NCHL/CCHL)*(TEM22*CCHL*C1-GZ*C2)
    * Death Rate of Phytoplankton
    SR33=NCHL*TEM11*C1
    *SINKS
    *Hydrolization of Organic Nitrogen
    SK3=(K34C+K34T*STP)*C3
    SK3A=K33*C3*[illegible]
    *Kinetic Term
    C33=SR31+SR34+SR32+SR33-SK3-SK3A
```

Where: SR31 = source term for organic nitrogen from decomposing dead zooplankton

SR34 = source term for organic nitrogen from decomposition of unassimilated herbivores zooplankton grazed by the 2nd level carnivores

SR32 = source term due to decomposing of unassimilated algae grazed by the herbivores zooplankton

SR33 = source term due to decomposing algae dead through endogenous respiration

SK3 = rate of hydrolysis to ammonia

SK3A = rate of sinking associated with settling organic matter

C3 = equilibrium concentration of organic nitrogen in mg N/l

C33 = kinetic concentration of organic nitrogen in mg N/l

```
(4) *System 4 Ammonia Nitrogen
    S44=SK3
    PREF=C4/(C4+C5)
    *SINKS
    SK41=GPC1*NCHL*PREF
    *Nitrification
    SK42=(K45C+K45T*STP)*C4
    *Kinetic Term
    C44=S44-SK41-SK42
```

Where: SK41 = conumption rate by growing phytoplankton

SK42 = loss through oxidation to nitrate

C44 = kintic concentration of ammonia nitrogen in mg N/l

```
(5)  *System 5 NO3
     SR5=SK42
     *Phytoplankton Uptke
     SK5=(1.-PREF)*NCHL*GPC1
     *Kinetic Term
     C55=SR5-SK5
```

Where: SK5 = consumption rate of growing phytoplankton (d^{-1})

C55 = kintic concentration in mg N/l

(6) The Particulate Phosphorus System

```
SR61=(PCHL/NCHL)*SR31
SR62=(PCHL/NCHL)*SR32
SR63=(PCHL/NCHL)*SR33
SR64=(PCHL/NCHL)*SR34
*SINK
SK6=(K67C+K67T*STP)*C6
SK6A=K66*C6*K4
*Kinetic Term
C66=SR63-SK6-SK6A+SR64+SR61+SR62
```

Where: SR61 = source of particulate phosphorus from dead, decomposing zooplankton

SR62 = source term due to decomposing, unassimilated algae grazed by the zooplankton

SR63 = source term due to decomposing algae dead through endogenous respiration

SR64 = source term due to decomposing, unassimilated zooplankton grazed by the higher level

SK6 = rate of oxidation to orthophosphate

SK6A = sinking rate associated with settling particles

C66 = kinetic concentration of particulate phosphorus

(7) The Orthophosphate System

```
SR7=SK6
SK7=PCHL*GPC1
SK7A=K77*C*K4
C77=SR7-SK7-SK7A
```

Where: SK7 = consumption rate by growing phytoplankton

SK7A = sinking rate associated with settling organic matter

C77 = kinetic concentration of soluble orthophosphate (mg P/l)

(8) The Carnivores Zooplankton

```
G8=A8Z*(CGC8+CGT8*STP)*C7
D8=K8+K8T*STP
C88=(G8-D8)*C8
```

Where: G8 = growth rate

D8 = death rate

The Differential Equations

```
C1DOT=(CO1/VOS)-(Q/VOS)*C1+C11
C2DOT=(Q/VOS)*(CO2-C2)+C22
C3DOT=(CO3/VOS)-(Q/VOS)*C3+(EA*ARES/VOL/DEPTH)*(C3S-C3)+
      C33+(WAORN/VOS)
C4DOT=(CO4/VOS)-(Q/VOS)*C4+(EA*ARES/VOL/DEPTH)*(C4S-C4)+
      C44+(WANH3/VOS)
C5DOT=(CO5/VOS)-(Q/VOS)*C5+(EA*ARES/VOL/DEPTH)*(C5S-C5)+
      C55+(CANO3/VOS)
C6DOT=(CO6/VOS)-(Q/VOS)*C6+(EA*ARES/VOL/DEPTH)*(C6S-C6)+
      C66+(WAORP/VOS)
C7DOT=(CO7/VOS)-(Q/VOS)*C7+(EA*ARES/VOL/DEPTH)*(C7S-C7)+
      C77+(WAPHO/VOS)
C8DOT=(Q/VOS)*(CO8-C8)+C88
```

Where: CnDOT = denotes the differential of the equilibrium concentration of the nth system with time

VOL = volume in m^3

Q = flow in 1/day

VOS = volume of the Main Body in 1/day

CnS = sediment concentration of the nth system

ARES = exchange surface area in m^2

COn = boundary loadings in mg or μg/day

n = 1,2,3,4,5,6,7

WANO3 = atmospheric nitrate nitrogen load in mg/day

WAORN = atmospheric organic nitrogen load in mg/day

WANH3 = atmospheric ammonia nitrogen load in mg/day

WAORP = atmospheric particulate phosphorus load in mg/day

WAPHO = atmospheric soluble orthophosphate load in mg/day

CO2,CO8= boundary concentration for the 2 zooplankton systems in mg C/1

TABLE 40. SURFACE AREA OF MAIN BODY

Height Above Sea Level (m)	Depth (m)	Area (m^2)	Percent Change in Area with Depth
132.4	0	3.8×10^7	100.0
132	0.4	3.8×10^7	99.5
130	2.4	3.7×10^7	97.5
128	4.4	3.5×10^7	92.1
126	6.4	3.2×10^7	84.2
124	8.4	2.5×10^7	66.1
122	10.4	1.5×10^7	40.8
120	12.4	5.0×10^6	13.2
119	13.4	0	0

TABLE 41. VOLUME OF MAIN BODY

Height Above Sea Level (m)	Depth (m)	Volume (m^3)	Volume Variation Percentage with Depth
132.4	0	0	0
132.2	0.4	3.6×10^4	0.01
131	1.4	1.1×10^6	0.3
129	3.4	8.1×10^6	2.3
127	5.4	2.4×10^7	6.9
125	7.4	7.5×10^7	21.4
123	9.4	1.7×10^8	47.1
121	11.4	2.8×10^8	80.9
119	13.4	3.5×10^8	100

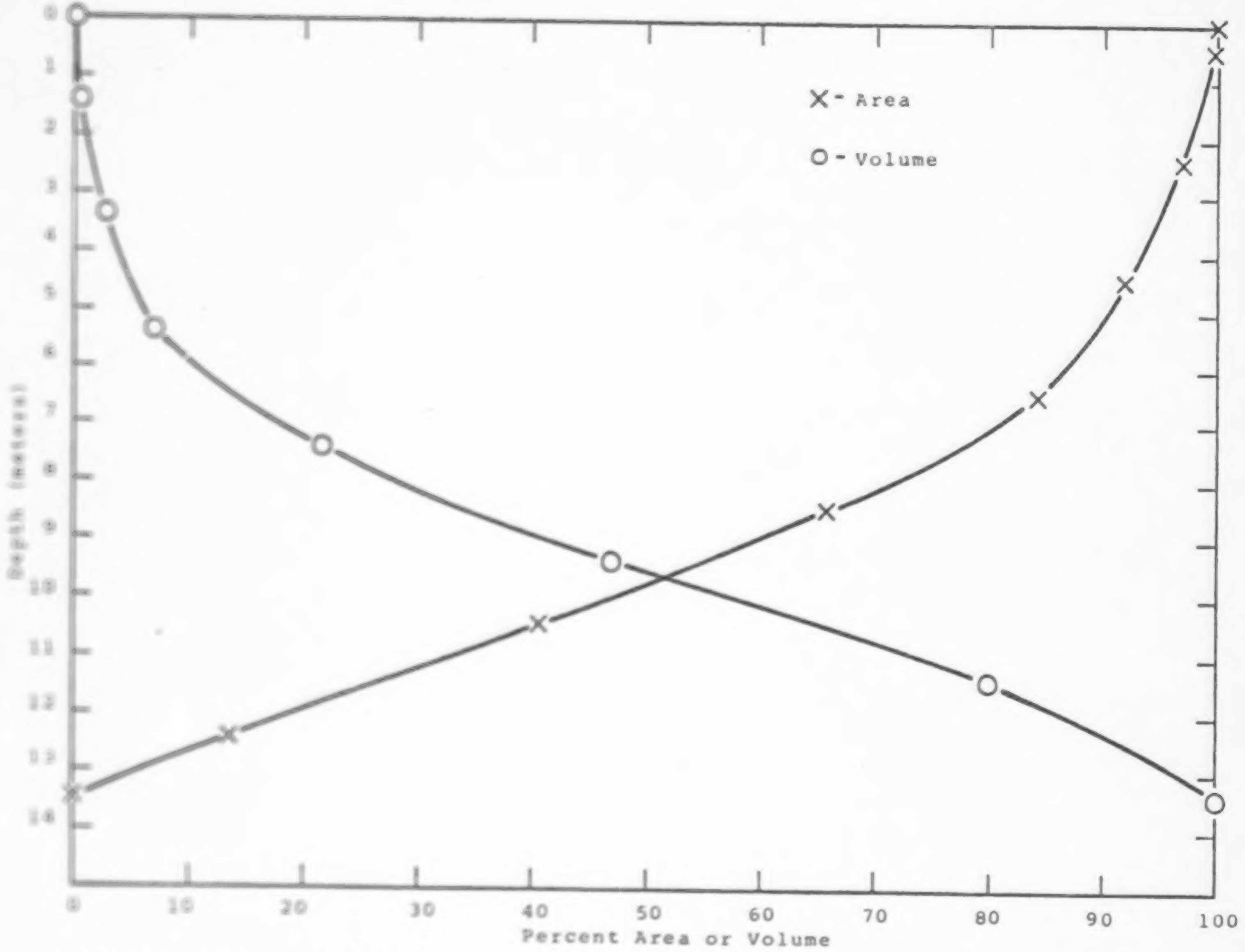

Figure 59. Hypsometric diagram - Main Body

Volume Input to the Model

There was a stable thermocline at a depth of 10 m in the Main Body from May through August which prevented mixing of hypolimnetic and epilimnetic waters. It was assumed that there would be no nutrient transport to the surface waters from the hypolimnion. During the summer months the behavior of the lake above the thermocline was modeled so the input volume to the model varied from winter to summer. Using the hypsometric diagram and assuming the thermocline was at 10 m throughout the summer months, the volume of the Main Body inputed to the model between September 15 and April 15 was 3.5×10^8 m^3, while during the summer months (May 1 to August 31) the volume was reduced to 2.02×10^8 m^3.

Hydrology

For the purposes of this model, it was assumed that the only water added to the lake was from the tributaries. Precipitation directly on the lake surface was not considered part of the input flow. This approach was used since, in general, it has been found that evaporation exceeds precipitation in the Dallas-Fort Worth region. Normally, Lake Ray Hubbard would receive 93.6 cm of rain per year directly on its surface. The estimated evaporation from this surface would be 208.8 cm.

The total flow input considered in the model was the sum of the three main tributaries, Rowlett Creek (Station 15 plus the Garland Sewage Treatment Plant flow at Station 24), Muddy Creek (Station 16), and Squabble Creek (Station 21 plus the Rockwall city sewage treatment plant flow). Table 42 presents the flows used for these computations. Table 43 presents the flow inputted to the model on a weekly basis. The weekly flows were determined by averaging the daily flows over a seven-day period.

Since the measurement of nutrient concentrations was terminated at the end of July 1977, it was necessary to estimate nutrient loads and flow inputs for the two weeks in August

TABLE 42. TOTAL TRIBUTARY FLOWS USED FOR ESTIMATED NUTRIENT LOADS TO LAKE RAY HUBBARD

Source	Flow (liters/day)
Rowlett Creek	5.21×10^7
Garland Sewage Treatment Plant	1.41×10^7
Muddy Creek	1.70×10^7
Squabble Creek	0.06×10^7
Rockwall Sewage Treatment Plant	0.15×10^7
Total	8.53×10^7

TABLE 43. TOTAL FLOW INPUT TO LAKE RAY HUBBARD

Date	Flow x 10^{-7} liters/day	Date	Flow x 10^{-7} liters/day
6/1/76	17.2	12/7	32.5
6/9	10.2	12/14	11.5
6/16	24.6	12/21	6.4
6/23	8.5	12/28	5.5
6/30	8.1	1/4/77	9.1
7/7	11.9	1/11	27.9
7/14	17.4	1/18	20.9
7/16	69.7	1/25	23.1
7/17	22.5	2/1	28.6
7/21	7.4	2/8	21.0
7/28	3.5	2/11	352.5
8/4	2.1	2/12	132.5
8/11	2.0	2/15	30.8
8/18	2.0	2/22	19.9
8/25	2.2	3/1	39.9
9/1	19.0	3/8	21.1
9/8	2.3	3/15	14.4
9/15	2.0	3/22	31.1
9/22	3.6	3/29	282.0
9/29	4.1	4/6	61.9
10/6	4.4	4/12	58.9
10/13	2.5	4/19	217.8
10/20	3.1	4/26	24.0
10/27	15.3	5/3	15.5
11/3	4.3	5/10	6.9
11/10	3.2	5/17	9.5
11/16	3.7	5/24	8.6
11/24	3.7	5/31	6.1
12/1	3.3	6/7	4.4

(continued)

Table 43 (Concluded)

Date	Flow x 10^{-7}liters/day
6/14	24.1
6/21	5.2
6/28	3.9
7/5	2.8
7/12	2.5
7/19	2.4
7/26	3.0
8/2	3.2
8/9	3.1
8/16	3.1

1977 when measurements were still being taken on the characteristics of the lake. The flow values used in the program during August 1977 were obtained by assuming an average July flow value of 2.92×10^7 liters/day for Stations 15 and 24 and by adding the hydrocomp computer estimates provided by the Dallas Water Utilities for Stations 16 and 2[illegible].

Temperature

Temperature is a basic driving variable in the model. Succession of zooplankton and phytoplankton groups due to temperature changes will be explored after the model has been calibrated. The average surface temperature of the Main Body ranged over the sampling period from a low of 4°C on 1/25/77 to 29.3°C on 8/16/77. Mean temperatures for the Main Body were obtained directly by averaging the surface temperatures for the four stations, 1, 2, 6, and 7. Table 44 shows average temperature values used as inputs to the model.

Light Intensity and Photoperiod

These values were taken from 1973 data collected by the Weather Bureau in Fort Worth (US Department of Commerce, 1973) and are presented in Table 45.

Extinction Coefficient Value

Measurements of light penetration with depth were made throughout the study period. The attenuation of light with depth in the Main Body was determined, and the extinction coefficient (Ke) was calculated. The extinction coefficient varied throughout the year, and this variability is reflected in the model by inputting measured Ke values for each sampling date on which light intensity measurements were made. The value used for the Main Body was obtained by averaging the light intensity at the same depth over the four stations located in the Main Body. Table 46 shows the average value of (Ke). Table 47 shows an example of the method for calculating the average extinction coefficient.

TABLE 44. AVERAGE TEMPERATURE FOR MAIN BODY

(°C)

Date	Average Temperature	Date	Average Temperature
6/8/76	24.6	5/31	26.4
6/15	24.4	6/14	26.6
6/23	24.9	6/28	27.3
6/29	27.1	7/12	28.0
7/6	26.8	7/26	29.3
7/13	26.9	8/2	29.2
7/20	27.2	8/9	28.0
7/27	27.0	8/16	29.3
8/3	28.2		
8/10	29.0		
8/17	28.3		
6/24	27.5		
8/31	27.6		
9/7	26.9		
9/14	24.9		
9/21	24.4		
9/28	24.2		
10/12	19.9		
10/26	16.2		
11/9	14.0		
11/30	8.5		
12/14	7.9		
1/25/77	4.0		
2/8	5.3		
3/1	11.1		
3/22	13.5		
4/12	17.6		
5/17	22.5		

TABLE 45. SOLAR RADIATION AND PHOTOPERIOD INPUT

Date	Solar Radiation (ly/day)	Photoperiod (days)
5/31/76	593	0.5746
6/15	632	0.5889
7/15	620	0.5924
8/15	573	0.5722
9/15	473	0.5354
10/15	387	0.4931
11/15	289	0.4535
12/15	237	0.4243
1/15/77	257	0.4188
2/15	324	0.4431
3/15	422	0.4775
4/15	480	0.5205
5/15	554	0.5604
6/15	593	0.5746
7/15	632	0.5889
8/15	620	0.5924

TABLE 46. AVERAGE EXTINCTION COEFFICIENT VALUES FOR THE MAIN BODY

Date	Ke(m^{-1})	Date	Ke(m^{-1})
6/29/76	0.76	1/25/77	1.24
7/6	0.84	2/8	1.14
7/13	1.03	3/1	1.33
7/20	1.09	3/22	1.64
7/27	0.81	4/12	1.64
8/3	1.14	5/3	1.50
8/10	0.94	5/17	1.34
8/17	1.03	5/31	1.13
8/24	1.20	6/14	0.85
8/31	1.11	6/28	0.88
9/7	1.28	7/12	1.33
9/14	1.28	7/26	0.70
9/21	1.40	8/9	1.08
9/28	1.70	8/16	0.95
10/12	1.66		

TABLE 47. EXAMPLE OF CALCULATION OF AVERAGE EXTINCTION COEFFICIENT

Station	Ke(m^{-1})	e^{-Ke}
1	0.741	0.477
2	0.866	0.421
6	0.519	0.595
7	0.955	0.385
		1.877

$$\frac{1.877}{4} = 0.469$$

$$\text{Average Ke} = \left| \ln 0.469 \right| = 0.756$$

On 6/29/76.

Nutrient Input

The sources of nutrients are runoff from natural drainage and urban developments and discharges from municipal wastewater treatment plants. The nitrogen and phosphorus loadings to the Main Body were assumed to be the sum of the loadings at Station 8 in Rowlett Arm, at Station 11 in Muddy Creek Arm, and at Station 3 in the East Fork. These loadings were calculated by multiplying the average weekly concentration at Station 8 (averaged over the water column) by the sum of the flows at Stations 15 and 24 on Rowlett Creek, the average weekly concentration at Station 11 by the flow at Station 16 on Muddy Creek, and the average weekly concentration at Station 3 by the sum of the flows at Squabble Creek and the Rockwall sewage treatment plant (estimated to be 1.514 x 10^6 liters/day). This approach was found to be more appropriate than using the loadings to Rowlett Arm and Muddy Creek Arm because Rowlett Arm, Muddy Creek Arm, and the East Fork Arm act as impoundments removing a substantial part of the phosphorus entering the Main Body of the lake. Most of the smaller tributaries were not flowing throughout most of the year, and therefore were not contributing significant amounts of nutrients to Lake Ray Hubbard. The loadings on each lake sampling date were inputted to the model. The CSMP program used at UTD is capable of interpolating linearly between measured input values to obtain a nutrient load at anytime, t. Table 48 presents the nutrient loadings inputted to the Lake I model.

Atmospheric Nutrient Inputs

The atmospheric nutrient loads were calculated by using the nutrient loading coefficients that were used for US OECD water bodies in the western US (Lakes Tahoe, Waldo, Sammamish, and Washington) (Rast and Lee, 1977). Table 49 presents the atmospheric loading coefficients used for estimating the atmospheric load to Lake Ray Hubbard. The total nitrogen load from the atmosphere was

TABLE 48 . NUTRIENT LOADINGS TO THE MAIN BODY INPUTTED TO THE MODEL

Date	Organic Nitrogen	Ammonia	Nitrate	Particulate Orthophosphate	Orthophosphate
6/08/76	7.97	3.16	19.18	2.17	4.65
6/15/76	7.47	9.30	4.02	2.32	5.10
6/23/76	7.78	1.43	5.46	0.93	2.20
6/29/76	5.34	0.34	0.79	0.68	1.36
7/06/76	3.50	0.53	0.49	0.31	1.37
7/13/76	7.33	1.97	2.16	2.42	8.83
7/20/76	2.53	0.31	0.94	0.59	2.44
7/27/76	1.60	0.04	0.07	0	0.01
8/03/76	1.41	0.13	0	0	0.02
8/10/76	1.50	0.05	0.04	0	0.02
8/17/76	1.13	0.10	0	0	0.23
8/24/76	1.87	0.05	0.13	0	0.06
8/31/76	1.37	0.20	2.65	0.13	1.44
9/07/76	1.53	0.05	2.19	0	0.01
9/14/76	1.10	0.03	0.62	0	0.06
9/21/76	1.30	0.03	0.34	0.40	1.42
9/28/76	1.21	0.03	0.39	2.45	7.43
10/12/76	1.52	0.11	0.27	1.47	4.67
10/26/76	1.29	0.04	13.00	8.50	27.36
11/09/76	1.44	0.30	1.31	0.80	2.42
11/30/76	1.50	0.25	0.81	2.09	6.37
12/14/76	0.95	0.04	5.57	9.24	28.77
1/25/77	1.14	1.15	34.09	32.69	100.90
2/08/77	0.85	0.04	102.30	18.64	59.30
3/01/77	6.40	2.19	171.90	16.61	63.60
3/22/77	158.10	1.33	104.10	27.84	56.50
4/12/77	66.10	19.64	526.34	48.27	100.15
5/17/77	0.64	0.61	77.08	2.08	6.95
5/31/77	0.57	0.43	29.30	3.83	8.36
6/14/77	0.55	0.09	1.31	21.04	43.11
6/28/77	0.75	0.01	1.34	0.89	2.91
7/12/77	0.74	0	0	0.84	2.54
7/26/77	0.71	0.01	0	2.05	6.16
8/02/77	0.93	0.01	0.29	2.20	6.66
8/09/77	0.95	0.06	0	12.36	7.07
8/16/77	1.50	0.08	0.50	2.23	6.73

Units are kg/day as N or P.

TABLE 49. ATMOSPHERIC LOADING COEFFICIENTS

Source	Watershed Export Coefficient ($g/m^2/year$)
Total P	
Rainfall	0.02
Dry Fallout	0.08
Total N	
Rainfall	0.8
Dry Fallout	1.6

TABLE 50. ATMOSPHERIC NUTRIENT LOADINGS TO THE MAIN BODY

Nutrient	Loading (mg/day)
Organic Nitrogen	62.465×10^6
Ammonia Nitrogen	93.698×10^6
Nitrate	93.698×10^6
Particulate Phosphorus	9.360×10^6
Orthophosphate	1.040×10^6

estimated to be 9.12 x 10^7 g/yr using the Main Body surface area of 3.8 x 10^7 m^2. It was assumed that 25 percent of the total atmospheric nitrogen loading was organic N, 37.5 percent was ammonia and 37.5 percent was nitrate. The total phosphorus load attributed to rainfail and dry fallout was estimated to be 3.8 x 10^6 g/yr. It was assumed that 90 percent of the atmospheric phosphorus load was particulate P and 10 percent was soluble ortho P. Table 50 presents the atmospheric loadings inputted to the model.

Chlorophyll Loading

The chlorophyll loading was calculated in the same manner as the nutrient loadings: the concentrations at Stations 3, 8, and 11 were multiplied by the flows of Squabble, Rowlett, and Muddy Creeks, respectively. Table 51 presents the chlorophyll loadings inputted to the model and Table 52 presents an example of the method of calculating the loadings.

Sediment Layer Concentrations

The nutrient concentration of the thin benthos layer that feeds the water column through dispersion was taken to be the average of the bottom concentrations at the four stations during the fall, winter, and spring, and the average of the bottom concentrations at Stations 2 and 7 during the summer months. Table 53 presents the concentrations of this layer inputted to the model.

Initial Conditions

The system of differential equations is solved using a set of initial conditions. The initial conditions obtained by averaging the concentration values in the Main Body on the first sampling date are presented in Table 54. The value of 0.10 mg/l for the zooplankton system was estimated since no data were collected on zooplankton in this study. The initial condition values of each parameter used in the Lake Ray Hubbard version of the Lake I model are presented in Table 55.

TABLE 51. CHLOROPHYLL LOADING INPUTTED TO THE MODEL

Date	Chlorophyll Concentrations (μg/l) Station 3	Station 8	Station 11	Total Load (10^{-7}μg/day)
6/15/76	12	24	24	60.4
6/29	7	11	5	9.61
8/3	10	30	28	7.47
9/14	16	36	22	8.99
9/29	14	28	18	13.3
11/9	12	18	14	7.15
12/14	10	54	26	59.9
1/25/77	19	76	43	170.3
3/1	39	9	24	60.9
3/22	23	92	30	153.8
4/12	7	3	7	24.5
5/17	24	4	8	8.17
6/14	16	40	10	95.6
6/28	24	38	64	17.8
7/12	21	22	18	8.30
8/2	16	44	18	15.4
8/9	15	30	18	11.1

TABLE 52. CALCULATION OF CHLOROPHYLL LOADING

Station	Chlorophyll Concentration ($\mu g/l$)	Flow x 10^{-6} l/day	Loading x 10^{-6} μg/day
11	5	0.919	4.59
8	11	7.004	80.55
3	7	0.525	10.96
		Total	96.1

TABLE 53. AVERAGE NUTRIENT CONCENTRATION IN THE SEDIMENT LAYER OF THE MAIN BODY INPUTTED TO THE MODEL

Date	Organic Nitrogen	Ammonia	Nitrate	Particulate Orthophosphate	Orthophosphate
6/08/76	2.48	0.07	0.08	0.02	0.02
6/15/76	3.18	0.06	0	0.04	0
6/23/76	3.43	0.22	0.50	0.69	0.01
6/29/76	2.40	0.03	0.01	0.09	0
7/06/76	1.34	0.03	0.07	0.03	0
7/13/76	0.86	0.04	0	0.04	0.01
7/20/76	1.19	0.04	0.18	0.05	0
7/27/76	0.84	0.02	0	0.05	0.01
8/03/76	0.93	0.03	0.02	0.05	0
8/10/76	0.81	0.04	0.01	0.05	0
8/17/76	0.84	0.05	0	0.06	0
8/24/76	0.98	0.02	0	0.06	0
8/31/76	0.91	0.07	0.01	0.05	0.01
9/07/76	1.11	0.31	0.03	0.09	0.01
9/14/76	0.78	0.14	0.01	0.04	0
9/21/76	1.33	0.03	0.01	0.02	0
9/28/76	0.63	0.02	0.01	0.05	0
10/12/76	1.05	0.05	0.05	0.03	0.01
10/26/76	0.89	0.04	0.07	0.03	0.02
11/09/76	2.42	0.17	0.06	0.04	0
11/30/76	0.97	0.07	0.02	0.03	0
12/14/76	1.06	0.02	0.03	0.04	0.01
1/25/77	0.63	0.03	0.07	0.04	0
2/08/77	0.41	0.04	0.12	0.03	0.01
3/01/77	0.48	0.15	0.04	0.04	0
3/22/77	0.97	0.01	0.14	0.12	0.01
4/12/77	1.35	0.08	0.25	0.02	0.01
5/17/77	0.29	0	0.51	0.03	0.01
5/31/77	0.51	0.09	0.33	0.07	0.02
6/14/77	0.47	0.01	0.12	0.04	0
6/28/77	0.44	0	0	0.05	0.01
7/12/77	0.45	0.06	0	0.03	0.01
7/26/77	0.47	0.03	0	0.05	0.01
8/02/77	0.52	0	0.01	0.04	0.01
8/09/77	0.54	0.04	0.01	0.06	0.01
8/16/77	0.54	0.02	0.01	0.05	0.01

Units are mg/l as N or P

TABLE 54. INITIAL CONDITIONS

System	Initial Conditions
Phytoplankton	8.5 µg/l
Zooplankton herbivorous	0.1 mg/l
Nitrogen	
Non-living organic	2.29 mg/l
Ammonia-N	0.08 mg/l
Nitrate-N	0.04 mg/l
Phosphorus	
Non-living particulate	0.022 mg/l
Available P	0.02 mg/l
Zooplankton carnivorous	0.005 mg/l

TABLE 55. INITIAL RUN PARAMETER VALUES

System - Parameter	Symbol	Value	Units
Phytoplankton			
Chlorophyll a	K1T	1.066	None
1. Saturated Growth Rate (0_0)	K1C	0.580	day^{-1}
2. Saturating Light Intensity	IS	350	ly/day
3. Michaelis Constant for Nitrogen	KMN	0.025	mg N/l
4. Michaelis Constant for Phosphorus	KMP	0.002	mg P/l
5. Endogenous Respiration Rate (0_{20})	K2T	1.08	None
	K2C	0.10	day^{-1}
6. Carbon to Chlorophyll Ratio	CCHL	0.05	mg C/μg Chl$_a$
7. Sink Velocity	SVEL	0.10	m/day
8. Preference Structure for Inorganic Nitrogen	PNH_3	0.5	None
9. Vertical Dispersion Coefficient	EA	777.6	m^2/day
Non-living Organic Nitrogen			
1. Organic N - NH_3 Hydrolysis Rate	K34T	0.00175	(day ^{o}C)$^{-1}$
(Decomposition Rate)	K34C	0	day^{-1}
2. Nitrogen to Chlorophyll a Ratio	NCHL	0.010	mg N/mg Chl$_a$
3. Sink Coefficient	K33	0.001	day^{-1}
Ammonia Nitrogen			
1. NH_3-NO_3^- Nitrification Rate	K45T	0.002	(day ^{o}C)$^{-1}$
	K45C	0	day^{-1}

Table 55 (Concluded)

System - Parameter	Symbol	Value	Units
Nitrate Nitrogen - none			
Non-living Organic Phosphorus			
1. Organic P - Inorganic P Conversion Rate	K67T K67C	0.007	(day °C)$^{-1}$
2. Phosphorus to Chlorophyll *a* Ratio	PCHL	0.001	mg P/μg Chl$_a$
3. Sink Coefficient	K66	0.001	day^{-1}
Available Phosphorus			
1. Sink Coefficient	K77	0	day^{-1}
Herbivorous Zooplankton-Carbon			
1. Grazing Rate	CGC	0	1/mg C/day °C
	CGT	0.06	1/mg C/day
2. Michaelis Constant for Phytoplankton	KMPL	30	μg Chl$_a$/1
3. Conversion Efficiency	AZP	0.60	None
4. Endogenous Respiration Rate	K4T	0.001	(day °C)$^{-1}$
	K4C	0.0	day^{-1}
Carnivorous Zooplankton-Carbon			
1. Grazing Rate	CGT8	0.06	1/mg C/day °C
	CGC^8	0	1/mg C/day
2. Conversion Efficiency	A8Z	0.60	None
3. Endogenous Respiration Rate	K8T	0.001	(day °C)$^{-1}$
	K8	0	day^{-1}

After Thomann, *et al*. (1975)

Reprogramming of Lake I Model

The first phase of the Lake Ray Hubbard modeling effort was undertaken by the City of Dallas Water Utilities, where members of the modeling group at DWU modified the Lake I model from CDC to IBM language. This in itself represents a significant contribution since one of the major drawbacks to utilization of the Lake I model by other investigators throughout the US was the fact that it was not in a language that could be readily used on the computers commonly available. This effort was accomplished by Larry Brown and Phyllis Cox under the supervision of Dan Brock of the Dallas Water Utilities.

The second phase of the DWU work was preparation of a user's manual. This manual was prepared by Phyllis Cox. This too represents a significant contribution since the Manhattan College group never prepared a user's manual for the Lake I model. It should be noted that the Manhattan College group did provide considerable assistance to DWU and UTD in connection with these efforts.

Continuous System Modeling Program-CSMP--

An important modification of the Lake 1 model program by the UTD group is the use of Continuous System Modeling Program (CSMP) (IBM, 1972). The CSMP is a computer package designed for simulation of continuous processes on a digital computer. This program was chosen for use with the Lake 1 model for the following reasons:

a) Ease of data input. User-oriented input capabilities permit input of discrete data. The computer program interpolates (using either a linear or quadratic fit) between points;

b) Superior output capabilities. User has complete control of graphic output including specification of output interval and which variables are to be plotted separately, which together;

c) Ease of experimental simulations. A variety of model simulation trials and systems analyses are easily accommodated;

d) Familiarity. The staff and students involved with modeling at UTD are completely familiar with the program. Thus, "debugging" problems were minimal.

The Lake I model (i.e. equation form, parameters set, etc.) is completely retained. In essence, the Lake I model has been translated from its original version into a language more amenable to simulation applications.

The Sections of the Lake I Model Program

A listing of the Lake I program as applied to Lake Ontario or Lake Ray Hubbard is available from the Center for Environmental Studies at UTD. The programs are coded in CSMP; however, familiarity with CSMP is not essential for understanding this discussion.

Section 1--

This section establishes the input values and physical parameters specific to Lake Ray Hubbard. Input temperature is provided through FUNCTION STP1. Paired values in this function represent time of year (in days starting June 1, 1976 as day number 1) and temperature value in °C at the beginning of each month. Similarly, other input functions are initiated in this section. These functions are: FUNCTION IAVI - for the surface average light intensity; and FUNCTION F1 - for the photoperiod. The surface area (AR) and the volume (VOZ) are treated constant and set in this section. The initial conditions for the eight modeled systems are given in this "initial" section of the program.

Section 2--

Section 2 initiates the time function with June 1, 1976 as day 1 in the Lake Ray Hubbard model, and interpolates linearly between the input functions.

The independent variable time is not explicitly available from CSMP. However, it may be accessed by generating a ramp of slope 1 which increases directly as time. After 365 days, a new ramp is generated. Although computer time continues to increase, model time is reset to 0. These operations are carried out by:

```
     B  = 0
     T  = RAMP(B)
     IF (T.GT.359.5) B = B + 360
```

The statement STP = AFGEN(STP1,T) is the linear interpolation command in this program. All input functions are similarly linearly interpolated and stored in the program in the same way.

Section 3--

Section 3 contains all the biological and chemical constants including the nutrient loadings W83, W84, W85, W86, and W87 in kg/day which were introduced at a constant value in Lake Ontario.

Section 4--

This section includes the conversions to proper units.

Sections 5 through 12--

These sections contain the kinetic equations for the phytoplankton chlorophyll, the herbivorous zooplankton, the organic nitrogen, the ammonia nitrogen, the nitrate nitrogen, the particulate phosphorus, the soluble orthophosphate, and the carnivorous zooplankton systems.

Section 13--

This section contains the differential equations used in the model.

Section 14--

The differential equations are integrated to yield a value for each state variable. Note the use of initial condition values and CSMP function INTGRL allows numerical integration by a user specified method.

Section 15--

User defined output is defined and the integration method is specified. Integration step size (DELT) is defined, output frequency (OUTDEL), and time of run (FINTIM) are given.

LAKE I MODEL RUN

It was originally planned to use previously collected data on Lake Ray Hubbard for the development and calibration of the model; the data collected during this study was to be used for verification of the model. After this study was initiated, it was found that the previously collected data on key parameters needed for model development were unreliable. As a result, the data collected during the current study had to be used for model development. Several problems developed with this approach. Shortly after initiation of this study, the City of Plano began to operate its new domestic wastewater treatment plant. This plant, in addition to providing conventional secondary biological treatment, also included coagulation with alum and effluent filtration. These two steps could lead to a significant reduction in the phosphorus content of the domestic wastewaters of Plano. Normally, a lake's response lags behind the change in its phosphorus load. As discussed by Sonzogni _et_ _al_. (1976), normally about three times the phosphorus residence time of the water body is needed to achieve approximately 90 percent of equilibrium for a new phosphorus load. Since the phosphorus residence time for Lake Ray Hubbard is about 10.7 months, it would take approximately 2.5 years for the lake to adjust to the new phosphorus load. Therefore, it is possible that the phosphorus load to Lake Ray Hubbard was changing during the course of the study, especially during the summer of 1977. Unfortunately, the North Texas Municipal Water District, which operates the Plano sewage treatment plant, does not measure total P in the effluent of the Plano plant on a regular basis. Therefore, it was not possible to make a direct comparison between total P concentrations prior to and after the initiation of the new treatment processes.

There were limited data available on total P concentrations in the effluent; however, no measurements were made before September 1976 when the plant began operating. Based on four measurements made between January and August 1977, the effluent total P concentrations after initiation of the treatment processes ranged from 2 to 6 mg/l which are somewhat lower than the concentrations found in the Garland sewage treatment plant effluent during this study. There are insufficient data to determine if the new treatment processes were affecting the total P concentrations in the Plano sewage treatment plant effluent.

In an attempt to determine if there was a difference in the phosphorus load, the loads of Rowlett Creek at Highway 78 before the plant began operating during the fall of 1976 were compared to those during the remainder of the study. No statistical difference was found at the 95 percent confidence level in the phosphorus loads prior to and after initiation of the treatment processes.

The Lake I model, as applied to the Main Body of Lake Ray Hubbard, is intended to simulate the effect of nutrients on the dynamics of phytoplankton biomass. The authors do not feel that the model is adequate at this time to accurately simulate response due to nutrient load alterations. Additional calibration is underway; however, no data are available for verification.

The initial simulation was obtained using a one segment version of Lake I. Values of the biological-chemical parameters were taken from the Lake I model calibrated for Lake Ontario (Thomann _et_ _al_., 1976) when data from Lake Ray Hubbard were not available. Lake Ray Hubbard data were available for flows, morphology, temperature, nutrient inputs, light extinction, and boundary and initial conditions. This run was made in order to compare the simulated concentrations with average data for the four stations in the Main Body.

Initial results (Figure 70) for the phytoplankton chlorophyll system show the concentration of chlorophyll a to be very low for the whole year (less than 1 μg/l). From the initial condition value, phytoplankton decreased rapidly and failed to recover.

Calibration

The emphasis of calibration work was placed on chlorophyll a dynamics. In the differential equation

$$\frac{dC1}{dt} = \frac{CO1}{Vol} - \frac{Q}{Vol} * C1 + C11$$

the loading term $(\frac{CO1}{Vol})$ is negligible and does not significantly affect the value of C1; so, C11 (the kinetic term for the plankton system) is controlling the equilibrium concentration of chlorophyll. The three terms affecting chlorophyll loss (which could justifiably be altered) are settling velocity, zooplankton grazing, and respiration. Since these three terms are correlated to temperature, they decrease in winter, thus favoring a winter phytoplankton bloom. The settling velocity was changed to a low value of 0.02 m/day in winter and a high value in summer of 0.1 m/day (as in Lake Ontario). Such a change did not noticeably affect the phytoplankton concentration.

The endogenous respiration expression used for Lake Ontario is:

$$TEM11 = K2C + K2T^{(Temp-20^{o})}$$

This expression gives very high respiration rates when applied to Lake Ray Hubbard in summer and results in a phytoplankton kill. A change in the exponential to (Temp-30°) did not lower the endogenous respiration rate to the needed level.

An altered exponential expression was then developed for the endogenous respiration rate in the form:

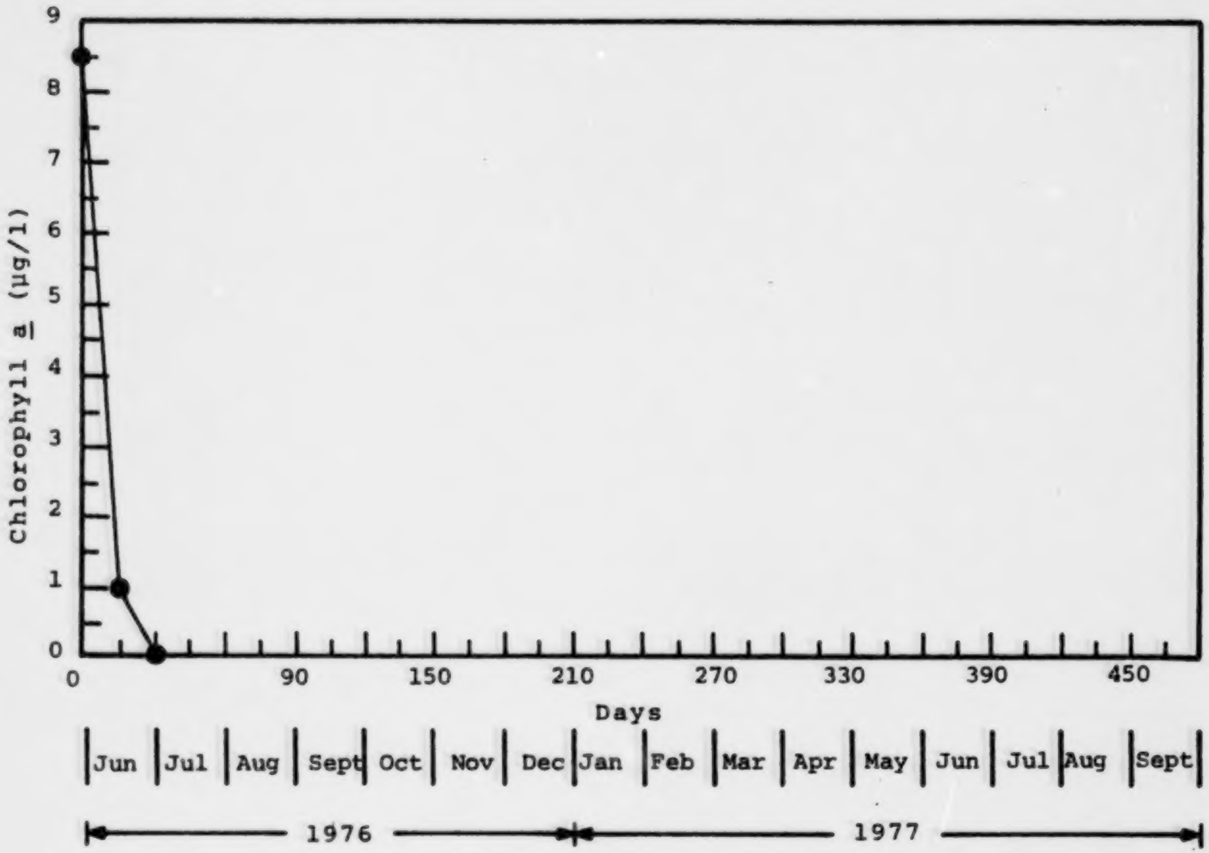

Figure 70. Initial run - phytoplankton chlorophyll a concentration

$$TEM11 = K2C * e^{K2T\,(Temp-30^{\circ})}$$

Figure 71 shows the new endogenous respiration rate function plotted against temperature. This function produces low rates of respiration below 10°C. (A sensitivity analysis will be done on the respiration rate term at a later time.) With the above changes made, the phytoplankton system showed three blooms during the study period, but the maximum chlorophyll a concentrations during simulated blooms were lower than the measured values.

To increase the chlorophyll a concentration, the Michaelis constants for inorganic nitrogen and for orthophosphate were decreased to allow less variability in the nutrient limitation terms affecting the growth of phytoplankton. The new values for KMN and KMP are 0.015 and 0.0008 mg/l, respectively. This change resulted in unreasonably high values of chlorophyll during the summer, but reasonable values otherwise.

Since zooplankton grazing is presumably higher in summer than at other times of the year, a sensitivity study which examined changes in the value of CGT (grazing coefficient) within the range of 0.06 to 0.25 was conducted. A value of 0.22 was found adequate during the summer while a value of 0.2 was reasonable for the winter. Figures 72, 73, and 74 present the phytoplankton chlorophyll system for three values of CGT.

In Figure 73, the modeled values for phytoplankton chlorophyll are plotted against the average measured values in the Main Body. The model output values are within the range of values found in summer but are generally lower than the measured values during winter. The modeled chlorophyll decreased too early in summer which could indicate that the rate of sinking out of the system was programmed too high at this time. The magnitude of the simulated winter bloom was lower than that from observed data. This may be due to the grazing rate in the model being lower than the actual grazing rate. Both the simulated and the actual data show low concentrations in the spring and blooms in the summer of 1977.

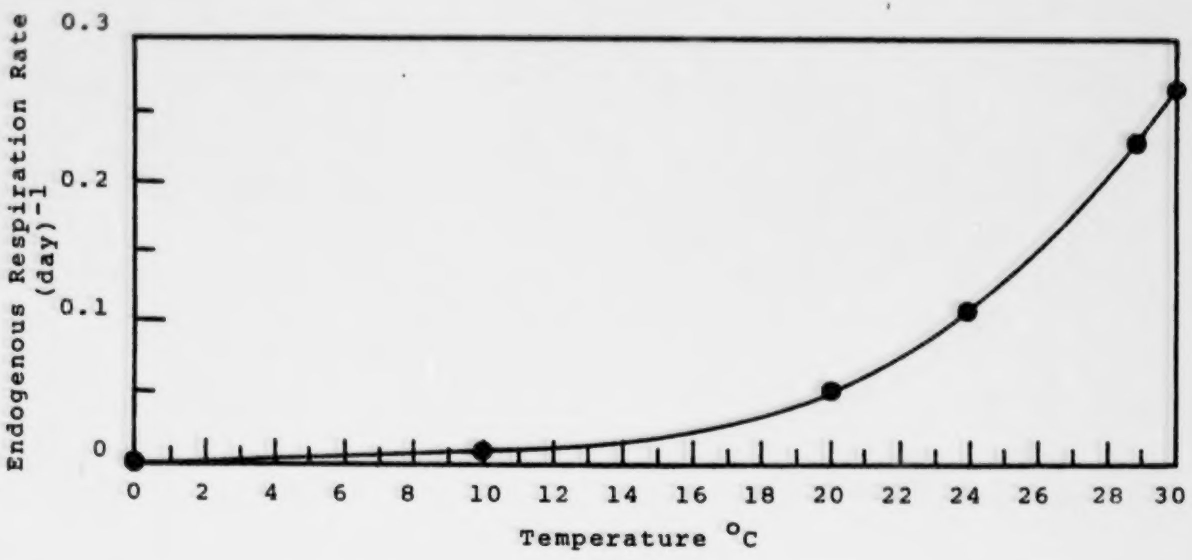

Figure 71 . Effect of temperature on endogenous respiration rate of phytoplankton population

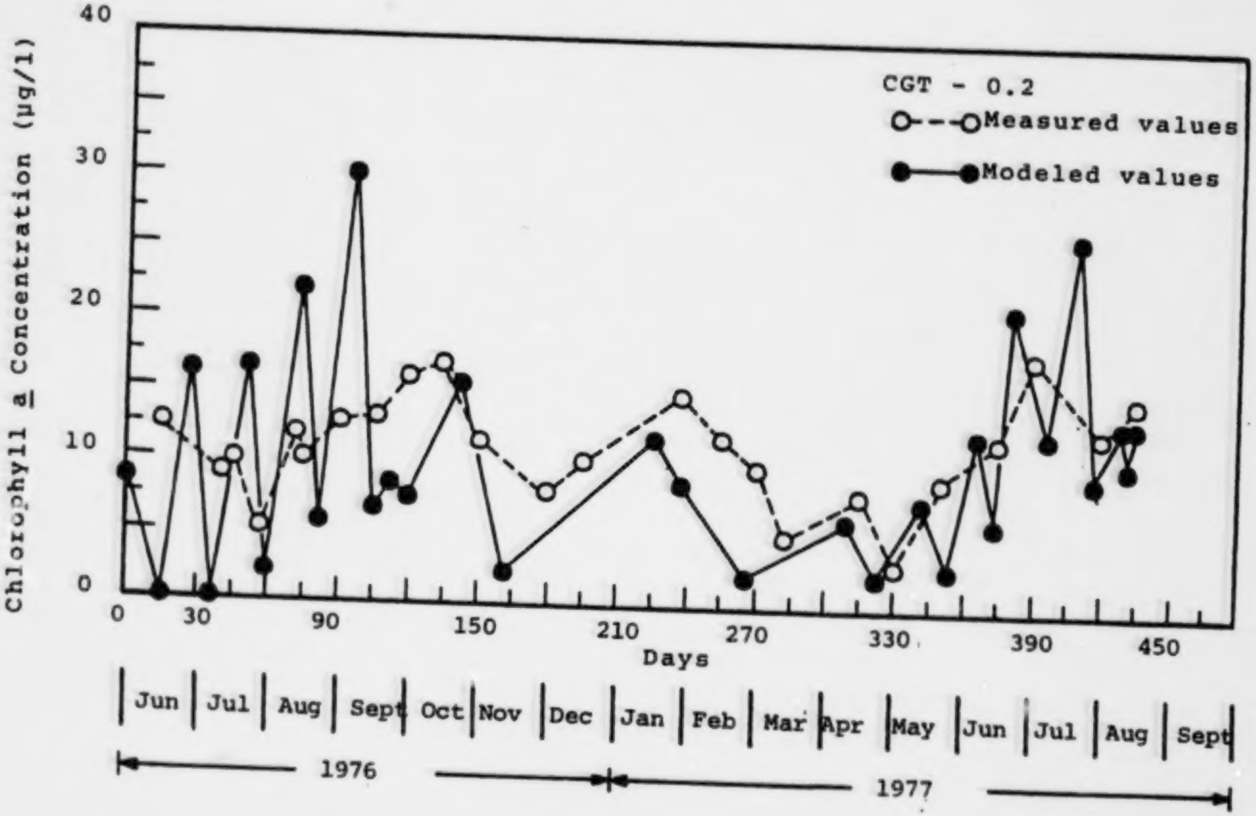

Figure 72 Comparison between modeled and measured chlorophyll a values (CGT = 0.2).

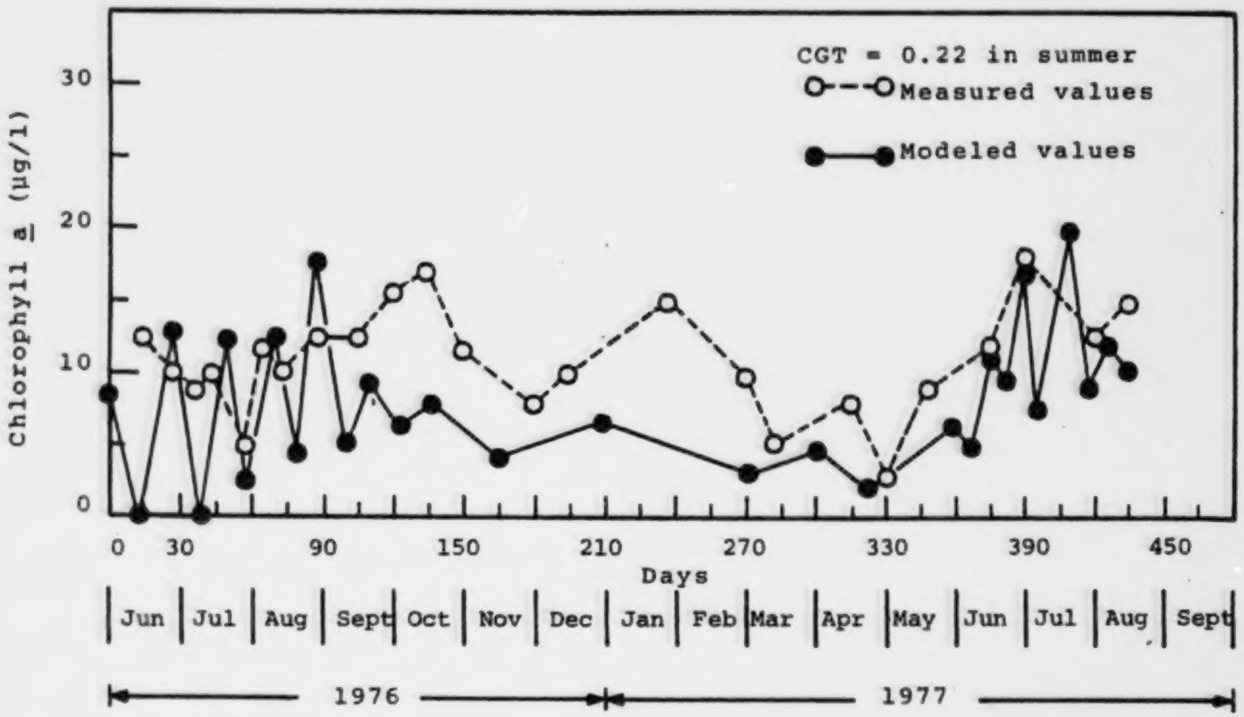

Figure 73 Phytoplankton chlorophyll system
Comparison between modeled and measured
chlorophyll a values (CGT = 0.22 in summer)

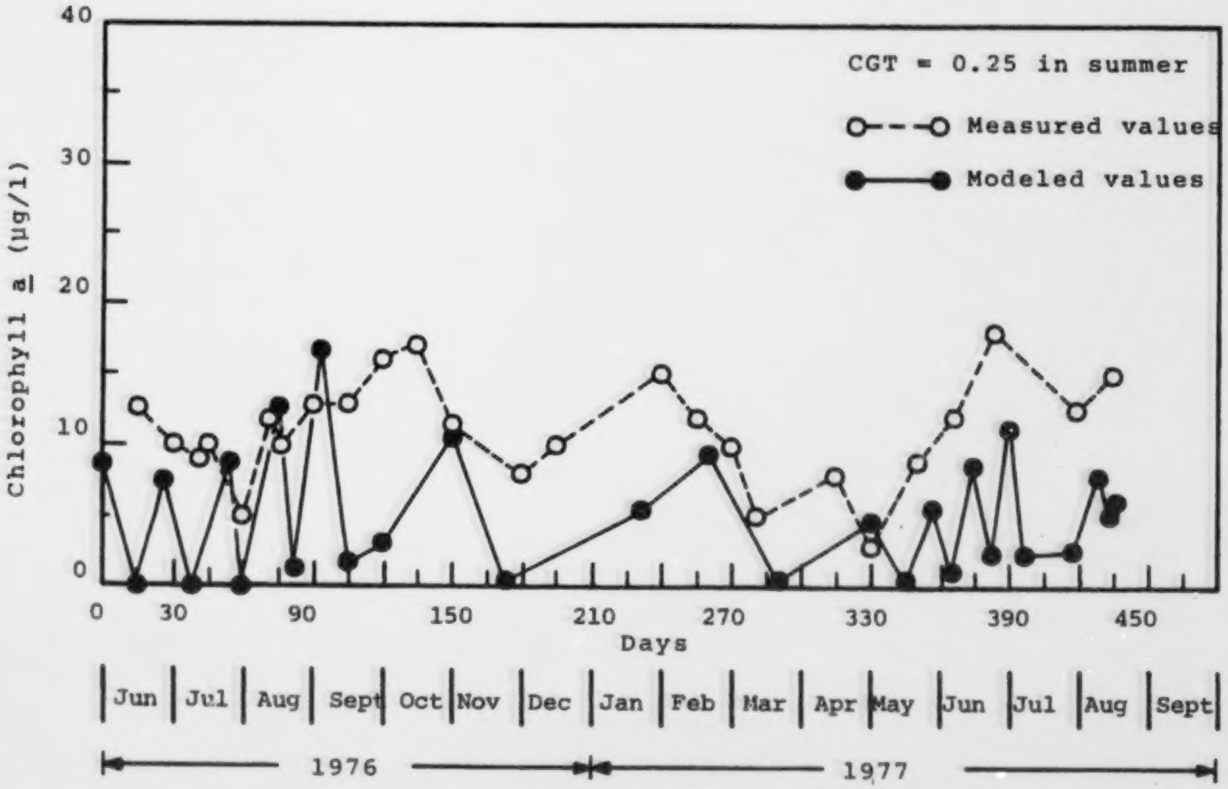

Figure 74. Phytoplankton chlorophyll concentration
Comparison between modeled and measured chlorophyll a values (CGT = 0.25 in summer)

Figure 75 presents the organic nitrogen system. The modeled and measured values were essentially equal during the entire study period. Figure 76 shows simulated and actual inorganic nitrogen concentrations during the study period. The fit is again quite good over the entire study period. Figures 77 and 78 present comparisons of the measured total phosphorus and soluble orthophosphate concentrations to simulated values. There is a reasonably good fit between measured and simulated values, but the modeled values lagged behind the actual values. The range of concentrations is the same for simulated and measured values.

It is felt that with additional effort, it will be possible to adjust the modified Lake I model so that it tracks, to a reasonable degree, both chlorophyll a and nitrogen and phosphorus compounds during the study period.

APPLICATION OF LAKE I RESULTS FOR DEVELOPMENT OF MANAGEMENT STRATEGIES

The primary purpose of developing a nutrient load-response model of the Lake I type is to develop a tool for predicting the response of the lake to an altered nutrient load. It became obvious shortly after initiation of this study that the key factor in model development for this purpose would not be attainable; namely, the verification of the Lake I model. Unfortunately, the previously collected data on Lake Ray Hubbard were of insufficient quantity and, in some cases, quality, to be used for calibration of the Lake I model. Attempts to calibrate the model to the conditions found in Lake Ray Hubbard were restricted to a five-week period from the end of data collection and sample analysis in early September 1977 until mid-October.

It would have been remarkable if Lake I had worked for Lake Ray Hubbard without major modifications. After the initiation of the study, it was found by Richardson and Bierman (1976) that the Lake I model did not properly simulate algal population dynamics in Saginaw Bay of Lake Huron. Saginaw Bay

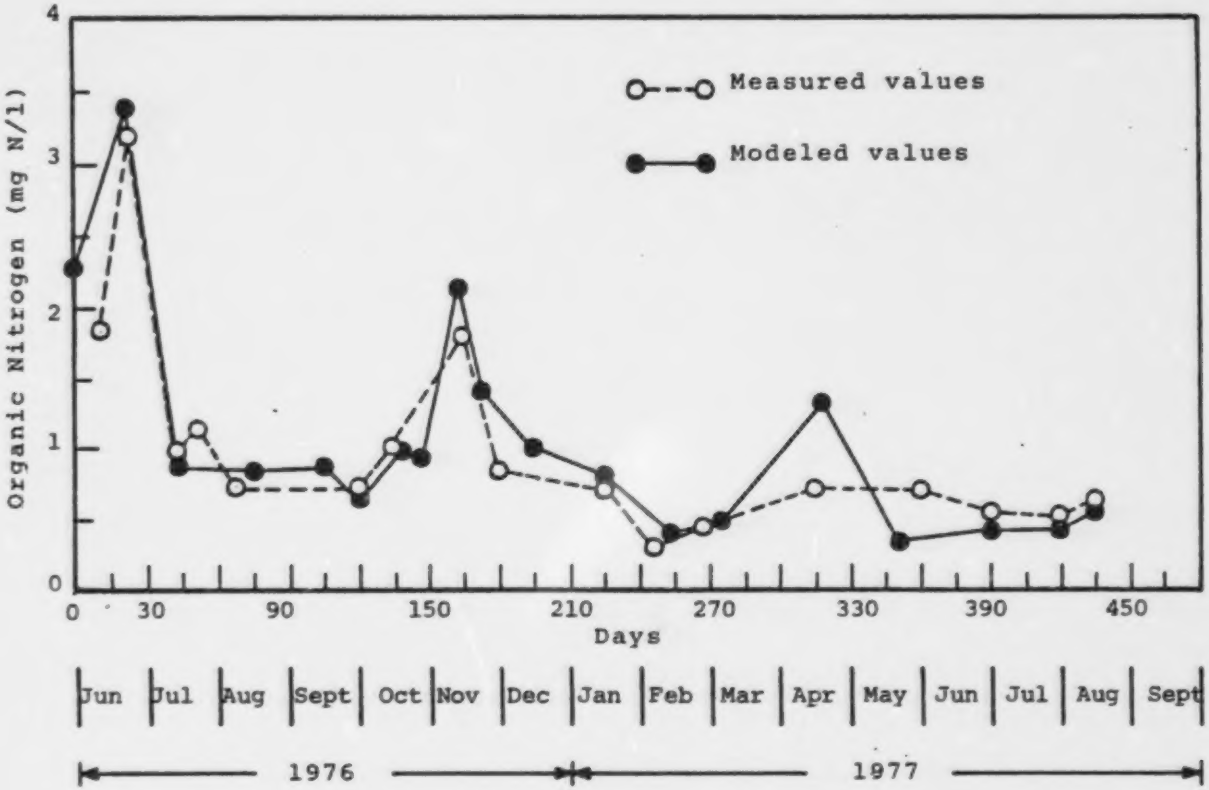

Figure 75 . Organic nitrogen system

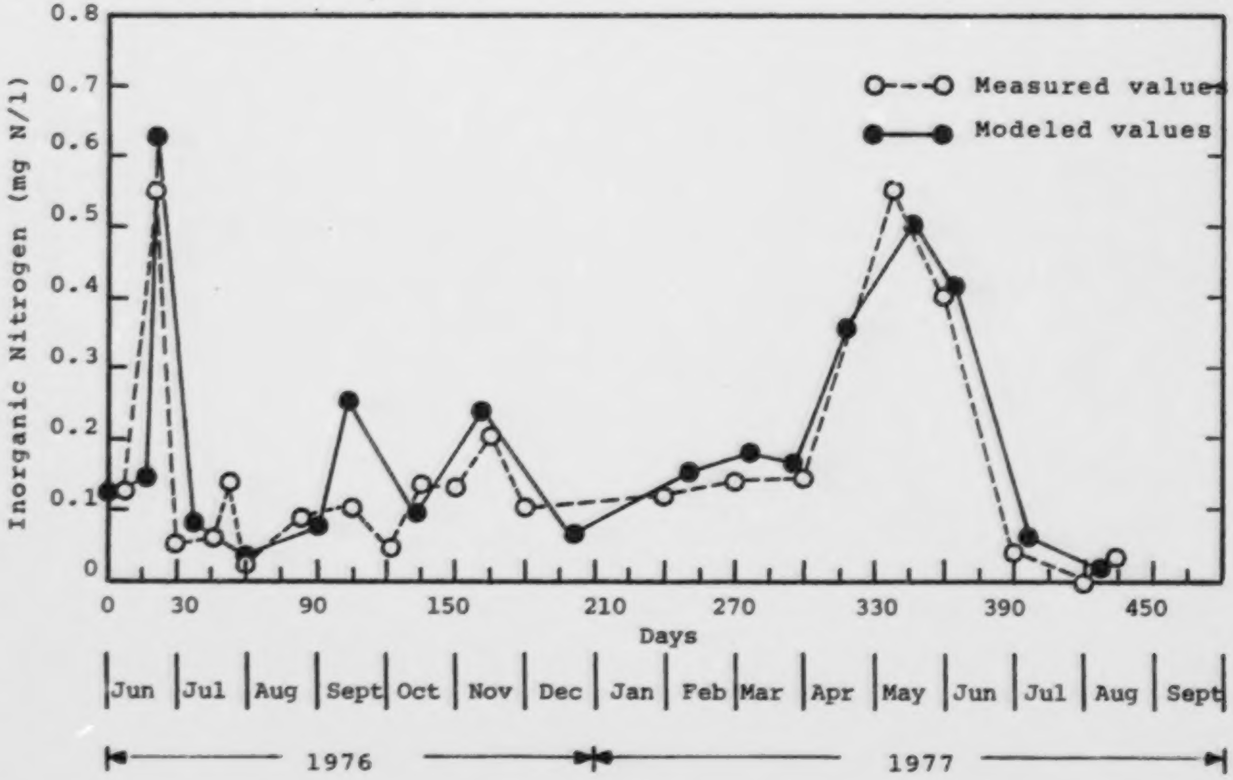

Figure 76 . Inorganic nitrogen

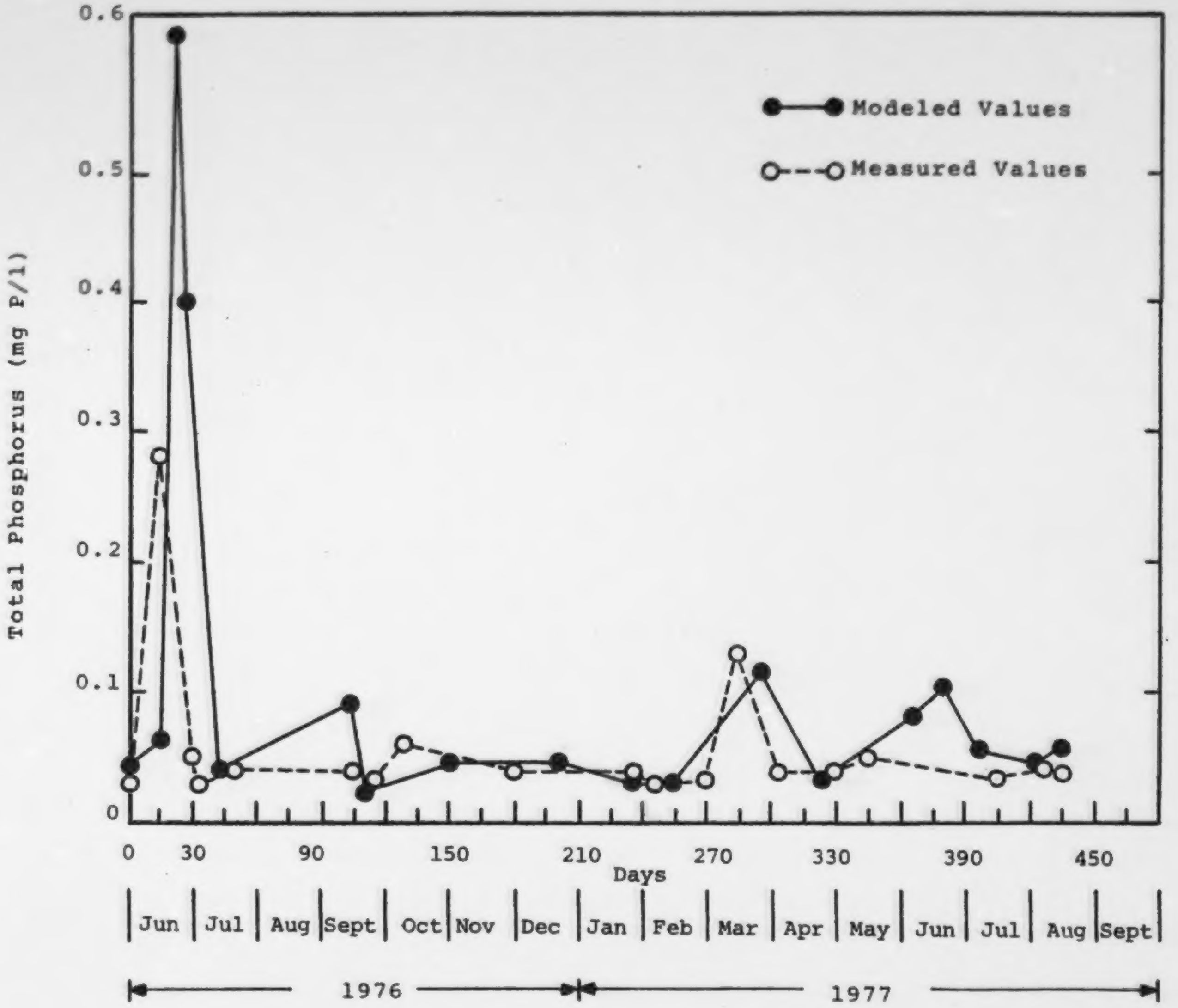

Figure 77. Total phosphorus

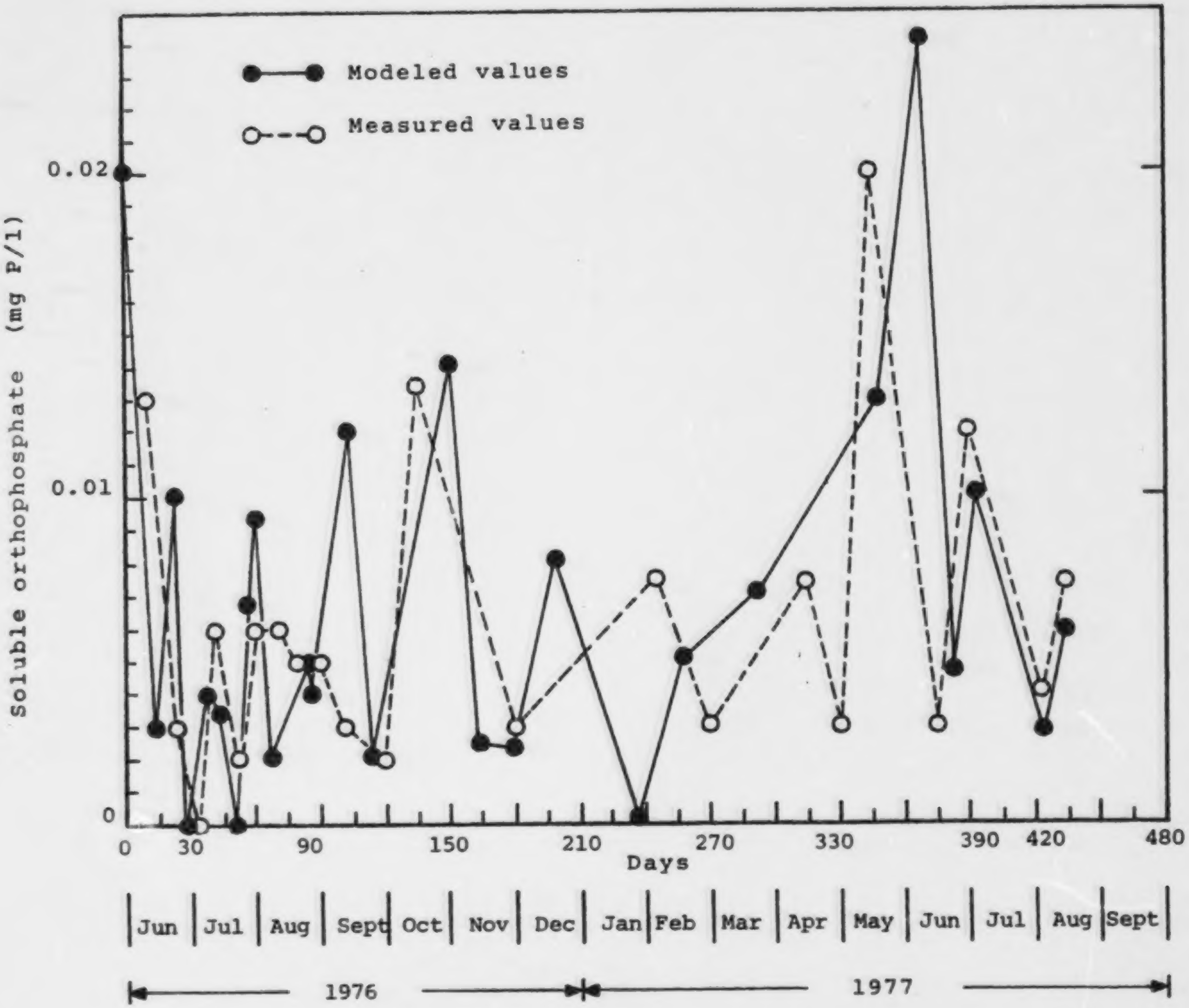

Figure 78. Soluble orthophosphate

is a shallow water body of high fertility which receives its nutrients from both domestic and land runoff sources. It has a number of properties which are very similar to those of Lake Ray Hubbard. If the model would not work for Saginaw Bay, without major modifications, then the likelihood of its working for Lake Ray Hubbard is relatively small.

It is believed by the authors that it will be possible, with additional time, to tune a modified version of the Lake I model so that it will properly track chlorophyll and the various nutrients found in the lake. The question that must be asked is whether the tuned-calibrated Lake I model should be used to attempt to predict the chlorophyll levels that will result in Lake Ray Hubbard arising from various alternative management schemes such as those that have been investigated in the OECD Eutrophication Modeling section of this report. The answer to this question is that it should not be used since the model has not been properly verified. It is impossible, under the circumstances that exist today, to properly verify this model. The best that can be done, once it is calibrated for the Main Body of the lake, is to examine how well it predicts the characteristics of the various arms. If it is able to predict the nutrient and chlorophyll content of the various arms of the lake, with adjustments made in the physical input functions which relate to the different sizes of the water bodies and the different flows, then this would represent a partial verification of this model.

SECTION 8*

OECD NUTRIENT LOAD-RESPONSE MODELING

About five years ago, the Organization for Economic Cooperation and Development (OECD) initiated an international study devoted to nutrient load-eutrophication response relationships for lakes and impoundments. This study involves approximately 200 water bodies located in the US, Western Europe, Canada, Japan, Australia, and New Zealand. It is divided into four regional projects with the US and Canada constituting the North American Project. Within the US, approximately 40 water bodies or parts thereof were investigated to determine the relationship between nitrogen and phosphorus loads to a water body and the characteristics of the water, with particular emphasis given to planktonic chlorophyll as a measure of eutrophication response. G. Fred Lee, at the University of Texas at Dallas, had a contract with the US EPA to prepare a critical review and synthesis of the US OECD eutrophication water bodies' data from the respective investigators' reports. A review has been prepared by Rast and Lee (1977) covering this review and synthesis. The OECD eutrophication modeling effort has been shown to be the most reliable method available today for describing nutrient load-eutrophication response relationships in lakes and impoundments with a wide range of trophic conditions, from oligotrophic (unproductive) to eutrophic (productive).

The basic approach used in the OECD eutrophication modeling effort is that originally developed by Vollenweider (1975) and used by the EPA in their National Eutrophication Survey. Vollenweider (1976) derived a simple relationship subsequently relating the phosphorus load to a phosphorus limited water body to the chlorophyll produced in the water body. The key factor in the Vollenweider modeling approach is that the phosphorus loads to a water body must be normalized among water bodies based on the mean depth and hydraulic residence time of the water body. It has been known for some time (Rawson, 1955) that shallow lakes tend to have greater algal production than deep lakes. Further, it was Vollenweider (1975) who was able to show that the hydraulic residence time of lakes (i.e., filling time) was a key factor in influencing how a lake utilizes its nutrients. When lakes are grouped according to similar phosphorus loads, mean depths and hydraulic residence times, they are also found to have similar chlorophyll concentrations and Secchi depths. While Vollenweider's original relationship was based on a few lakes for which there was a relatively limited data base, Rast and Lee, working with a much larger data base, have shown that this approach is applicable to the US OECD eutrophication study water bodies. Similar results are being obtained in the other three regional projects of the OECD Eutrophication Program.

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*Walter Rast coauthored this section of the report.

Rast and Lee (1977) have expanded Vollenweider's basic relationship to include an estimation of the Secchi depth of a water body based on the phosphorus load and mean depth/hydraulic residence time characteristics. Their model assumes that the Secchi depth is controlled by phytoplankton growth and not by color or sediment-derived turbidity. Rast and Lee were also able to derive a relationship between the phosphorus load to a water body and its hypolimnetic oxygen depletion. For a more detailed discussion of these relationships, the development of the OECD modeling approach, and the results of the OECD eutrophication study, consult the report by Rast and Lee (1977).

In order to apply the OECD eutrophication modeling to Lake Ray Hubbard, it was necessary to compute certain characteristics of the lake, or parts thereof, to which the modeling effort is to be applied. Two parts of the lake have been considered for the application of the OECD eutrophication modeling approach, the Main Body and Rowlett Arm. Table 56 presents the characteristics of these two parts of the lake that are necessary for application of the OECD eutrophication modeling approach to Lake Ray Hubbard. The development of the values in this table has been discussed in other parts of this report. It is important to note that the Main Body phosphorus load used for these computations was based on the concentrations of phosphorus at Stations 3, 8, and 11, multiplied by the flows of Squabble Creek, Rowlett Arm, and Muddy Creek, respectively. It was not technically valid to use the total phosphorus loads to the lake in these computations since the arms of the lake remove a significant amount of the total phosphorus, making it unavailable for algal growth within the Main Body. Table 57 presents an explanation of the terms used in the normalized phosphorus load plots.

This section contains estimates of the characteristics of the Main Body and Rowlett Arm as a result of the various hypothetical manipulations of the phosphorus load, including advanced waste treatment for phosphorus removal at the sewage treatment plants, reduction of phosphorus load by a phosphate detergent ban, and through increased population in the watersheds of the tributaries of Lake Ray Hubbard. Further, consideration was given to the impact of normal flow from Lake Lavon and its associated nutrients on the Main Body of the lake. Since the nutrients enter the arms before reaching the Main Body, it was necessary to assume a certain retention coefficient under new nutrient loads, which could be derived as a result of various management practices. The retention coefficient defines the portion of the phosphorus load which is retained or sedimented in the lake (i.e., it is not lost by way of outflow). It is defined as $1/(1+\sqrt{\rho_\omega})$ (Vollenweider, 1975), where $\rho\omega$ = 1/hydraulic residence time, τ_ω. For example, with respect to Lake Lavon, the US EPA (1977a) estimated that approximately 20,000 kg of phosphorus left

TABLE 56. OECD EUTROPHICATION MODELING CHARACTERISTICS OF THE MAIN BODY AND ROWLETT ARM OF LAKE RAY HUBBARD

	Main Body	Rowlett Arm
Phosphorus Load (g P/yr x 10^{-6})		
Atmosphere	3.7	0.7
Tributaries		
1976-77	11.9	181.8
with 35% removal at STP's	10.0	150.0
with 85% removal at STP's	7.2	94.9
1976-77 load & Lake Lavon load	12.9	-
with 35% removal at STP's	11.0	-
with 85% removal at STP's	8.2	-
1990 estimated loads including Lake Lavon	18.4	290.7
with 85% removal at STP's	9.1	111.2
Inflow (l/yr)		
1976-77	1.0×10^{11}	7.0×10^{10}
with Lake Lavon flow	2.9×10^{11}	7.0×10^{10}
Surface Area (m^2)	3.8×10^{7}	6.8×10^{6}
Volume (l)	3.5×10^{11}	1.9×10^{10}
Hydraulic Residence Time-τ_ω (yrs)		
1976-77	3.5	0.27
with Lake Lavon flow	1.2	0.27
Mean Depth - $\bar{Z}$ (m)	9.2	2.8
Mean Summer Epilimnetic Chlorophyll a (µg/l) 1976	11.5	47.0
Mean Summer Secchi Depth (m) 1976	1.3	0.3

TABLE 57. PHOSPHORUS LOADING CHARACTERISTICS

Symbols used in Equation	Explanation of Symbol	Units	Method of Calculation
L(P)	Areal P loading	mg P/m^2/yr	$\frac{\text{P loading(mg/yr)}}{\text{Surface Area of Main Body or Rowlett Arm (m}^2\text{)}}$
$\bar{z}$	Mean Depth	m	$\frac{\text{Volume (m}^3\text{)}}{\text{Surface Area (m}^2\text{)}}$
τ_ω	Hydraulic Residence Time	yr^{-1}	$\frac{\text{Inflow (m}^3\text{/yr)}}{\text{Volume (m}^3\text{)}}$
q_s	$\bar{z}/\tau_\omega$	m/yr	$\frac{\text{Mean Depth (m)}}{\text{Hydraulic Residence Time (yr)}}$
P	P Retention Coefficient	%	$\frac{1}{1 + \sqrt{\frac{1}{\tau_\omega}}} \times 100$

this lake and entered Lake Ray Hubbard in 1974. For the purpose of modeling the eutrophication behavior of the Main Body, the amount of phosphorus actually reaching the Main Body had to be determined. For these calculations, it was arbitrarily assumed that 95 percent of the phosphorus discharged from Lake Lavon would be retained within the East Fork Arm. In other words, it was assumed that only 1,000 kg/yr of the 20,000 kg/yr phosphorus load from Lake Lavon would reach the Main Body of Lake Hubbard. It is likely, as the population in the Lake Lavon watershed increases, that the concentrations of phosphorus in Lake Lavon will also increase and with time Lake Lavon will be contributing increasing amounts of phosphorus to the Main Body of Lake Ray Hubbard.

It was necessary to assume in these computations that the retention coefficient of phosphorus in Rowlett Arm remained the same, regardless of whether the load was increased due to increased population, or decreased due to a detergent phosphate ban or phosphate removal by advanced waste treatment processes at the sewage treatment plants. This assumption is probably reasonable, based on the magnitude of the changes that could take place in the overall degree of fertility of the lake or its arms. It should be noted, however, that Rast and Lee (1977) have found that the retention coefficient of phosphorus as measured by the phosphorus residence time in water bodies seems to be inversely proportional to the degree of productivity of a water body. The most productive lakes generally tend to retain the greatest amounts, while lakes of low productivity retain very little phosphorus. This relates to the fact that one of the principal modes of phosphorus retention is its incorporation into algal cells, which can eventually carry it to the sediments. If there is a change in the degree of phosphorus retention within Rowlett Arm as a function of its phosphorus load, it would likely be toward a lower retention of phosphorus under lower loads and slightly greater phosphorus retention under higher loads. This assumes that phosphorus is, or can be made, limiting with respect to controlling algal growth in Rowlett Arm. It appears, as discussed previously, that under certain conditions this may not be the case. However, most of the time it appears that phosphorus is at or near concentrations which could control algal growth in Rowlett Arm.

Figure 79 presents a plot of the phosphorus loads to the Main Body and Rowlett Arm of Lake Ray Hubbard as a function of the mean depth/hydraulic residence time of these parts of the lake under the various conditions considered in this evaluation. An explanation of the symbols used in this and subsequent figures is given in Tables 57 and 58. Examination of this figure shows that the phosphorus load to the Main Body of the lake (point A) was 0.41 g $P/m^2/yr$ during the period from June 1976 to May 1977. This load placed this part of the lake in the excessively loaded area of the curve. As discussed by Rast and Lee (1977), this excessive loading line is based on impairment of recreational use as derived from studies in southern Wisconsin lakes in the mid-1940's. It has been found that, in general, the excessive loading zone in Figure 79 represents the trophic zone where an impairment of the use of the lake has occurred because of the excessive growths of algae. There would also likely be an adverse impact on the use of the water for domestic water supply purposes. As a result of having examined the relationships of the nutrient load-water quality characteristics of many water bodies across the US, Rast and Lee have determined that the position of the Main Body of Lake Ray Hubbard as shown in Figure 79 is appropriate, relative to other water bodies with similar phosphorus loads and hydrological and morphometric characteristics.

As shown in Figure 79, if 85 percent of the phosphorus in domestic wastewaters was removed by advanced wastewater treatment (point B), the phosphorus load to the Main Body would be reduced from 0.41 to 0.29 g $P/m^2/yr$. This is based on the inflow of 1.0 x 10^{11} l/yr found during the study period and assumed no flow from Lake Lavon. In order to estimate the potential significance of a detergent phosphate ban in the Lake Ray Hubbard watershed, a 35 percent removal of phosphorus from domestic wastewaters was

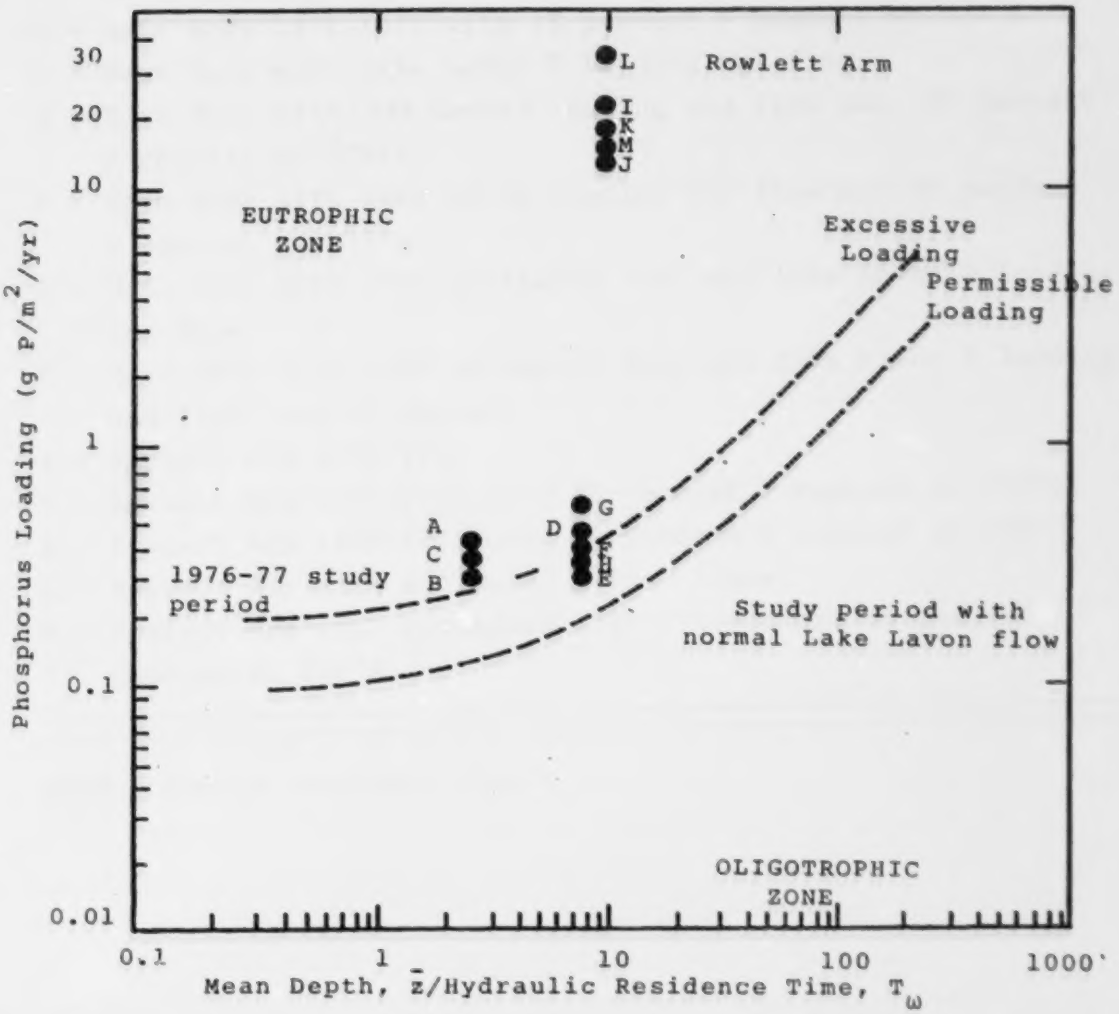

Figure 79. Phosphorus loading to the Main Body and Rowlett Arm of Lake Ray Hubbard

TABLE 58. SYMBOLS USED IN NUTRIENT LOAD-LAKE RESPONSE FIGURES

A = Main Body 1976-1977
B = Main Body 1976-1977 with 85 percent P removal at STP's.*
C = Main Body 1976-1977 with 35 percent P removal at STP's.
D = Main Body with Lake Lavon P loading and flow.
E = Main Body with Lake Lavon P loading and flow and 85 percent P removal at STP's.
F = Main Body with Lake Lavon loading and flow with 35 percent P removal at STP's.
G = Main Body with 1990 estimated load and Lake Lavon P loading and flow.
H = Main Body with 1990 estimated load and Lake Lavon P loading and flow and 85 percent
I = Rowlett Arm 1976-1977
J = Rowlett Arm 1976-1977 with 85 percent P removal at STP's.
K - Rowlett Arm 1976-1977 with 35 percent P removal at STP's.
L = Rowlett Arm with estimated 1990 P loads.
M = Rowlett Arm with estimated 1990 P loads and 85 percent P removal at STP's.

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*STP = Sewage treatment plants

assumed and the resultant phosphorus loading was calculated. This is shown as point (C) under the flow regimes that occurred during the study period. The load reduction arising from a detergent phosphate ban in the Lake Ray Hubbard watershed would be from 0.41 to 0.36 g $P/m^2/yr$.

Since the study period was atypical with respect to flows from Lake Lavon, and since the normal Lake Lavon flows would decrease the hydraulic residence time in the Main Body of Lake Ray Hubbard from 3.5 years to 1.2 years, it is of interest to estimate what the characteristics of Lake Ray Hubbard would have been during the study period if it were assumed that the normal Lake Lavon flow computed by the DWU had occurred during the study period and 1,000 kg phosphorus had entered the Main Body of the lake as a result of this flow. The 1976 phosphorus loadings assuming no P removal (D), 35 percent P removal (F), and 85 percent P removal (E) from domestic wastewaters are plotted in Figure 79 with the increased mean depth hydraulic residence time relationship due to the Lake Lavon flow. With normal P removal, the point is on the excessive loading line. With 85 percent P removal, the point is approximately halfway between the excessive and permissible loading lines; and with 35 percent removal, the point is just below the excessive loading line. As shown in Figure 79,the decreased hydraulic residence time causes the position of Lake Ray Hubbard on the Vollenweider diagram to shift significantly to the right so that it could receive a higher phosphorus load, yet maintain the same water quality as with the longer hydraulic residence time. The importance of the Lake Lavon flow in influencing the behavior of Lake Ray Hubbard cannot be over-emphasized. This is discussed further in a subsequent section of this report.

The 1990 projected load to the Main Body was calculated to be 0.58 g $P/m^2/yr$, assuming no attempt is made to control the phosphorus input to Lake Ray Hubbard from domestic wastewaters, and assuming that normal flows from Lake Lavon occur. This places the point (G) above the excessive loading line as shown in Figure 79 . However, if 85 percent P removal from domestic wastewaters is initiated, the phosphorus load to Lake Ray Hubbard in 1990 (point H) will actually be less than the 1976-77 load. As shown in Figure 79, with 85 percent P removal from domestic wastewaters, the phosphorus load to the Main Body was calculated to be 0.34 g $P/m^2/yr$, which places the point between the excessive and permissible loading lines.

It is evident from examination of Figure 79 that relatively large changes in phosphorus load to the lake can occur and have limited impact on the position of the Main Body on the Vollenweider loading diagram. Even with 85 percent phosphorus removal from domestic wastewaters, the Main Body would still be classified as eutrophic compared to other lakes, based on its 1976-77 phosphorus load and mean depth/hydraulic residence time characteristics, and would likely continue to have degraded water quality.

Examination of Figure 79 shows that the phosphorus load as a function of the mean depth/hydraulic residence time places Rowlett Arm well above the excessive loading line. The 1976-77 phosphorus load (point I), as determined in this study, was 26.8 g $P/m^2/yr$. This places Rowlett Arm among water bodies with the highest phosphorus loads studied by Rast and Lee as part of the US OECD Eutrophication Program review. Eighty-five percent removal of the phosphorus from domestic wastewaters reduces the P load to Rowlett Arm to 14.1 g $P/m^2/yr$. The 1990 estimated load (point L), assuming no removal of phosphorus from domestic wastewaters, is 42.8 g $P/m^2/yr$. If 85 percent of the phosphorus is removed from domestic wastewaters, the 1990 phosphorus load to Rowlett Arm (point J) will be 16.5 g $P/m^2/yr$ which is less than the current phosphorus load. This indicates that the water quality of Rowlett Arm can be maintained at its present level or possibly slightly improved by initiation of 85 percent removal of phosphorus from domestic wastewaters by advanced wastewater treatment. However, it is noted that Rowlett Arm still remains in a eutrophic position on the Vollenweider plot, regardless of the remedial measure used.

Figure 80 presents the relationship between the phosphorus loads as normalized by mean depth and hydraulic residence times, and the summer mean epilimnetic chlorophyll concentrations within the Main Body and Rowlett Arm of Lake Ray Hubbard. One of the most significant findings of this study is that the average summer epilimnetic chlorophyll concentration of the Main Body is only slightly lower than the concentration predicted by the model based on the phosphorus load (point A). These results indicate that the OECD phosphorus loading relationship is applicable to Lake Ray Hubbard. For the first time there is evidence that warm water bodies such as those that are found in the southern half of the US, show the same nutrient load-eutrophication response relationship as north temperate water bodies which have been extensively studied. These results also indicate that the approach used to estimate the phosphorus loads to the Main Body is reasonable. Concentrations at Stations 3, 8, and 11, multiplied by the flows of Squabble, Rowlett, and Muddy Creeks, respectively, as an estimate of the phosphorus load to the Main Body predicts the chlorophyll found in the Main Body. The third aspect which is of importance is that since this approach can predict the chlorophyll level under the current loading relationship, there is reason to believe that it would be valid in predicting the chlorophyll levels under other phosphorus loads which could result from certain types of management strategies. In order to predict the mean chlorophyll concentration for each hypothetical loading scheme, it was assumed that the line for the Main Body would be parallel to the OECD line and the same distance below the line as the 1976-77 point (A) is.

As shown in Figure 80, the average growing season epilimnetic chlorophyll a concentration in the Main Body during the study period (A) was 11.5 µg/l. With 85 percent removal of phosphorus from domestic wastewaters (B), this could be reduced to 7.5 µg/l and with 35 percent removal (C), it could be reduced to 10 µg/l. However, reductions of this magnitude would not produce a readily detectable change in water quality in the Main Body.

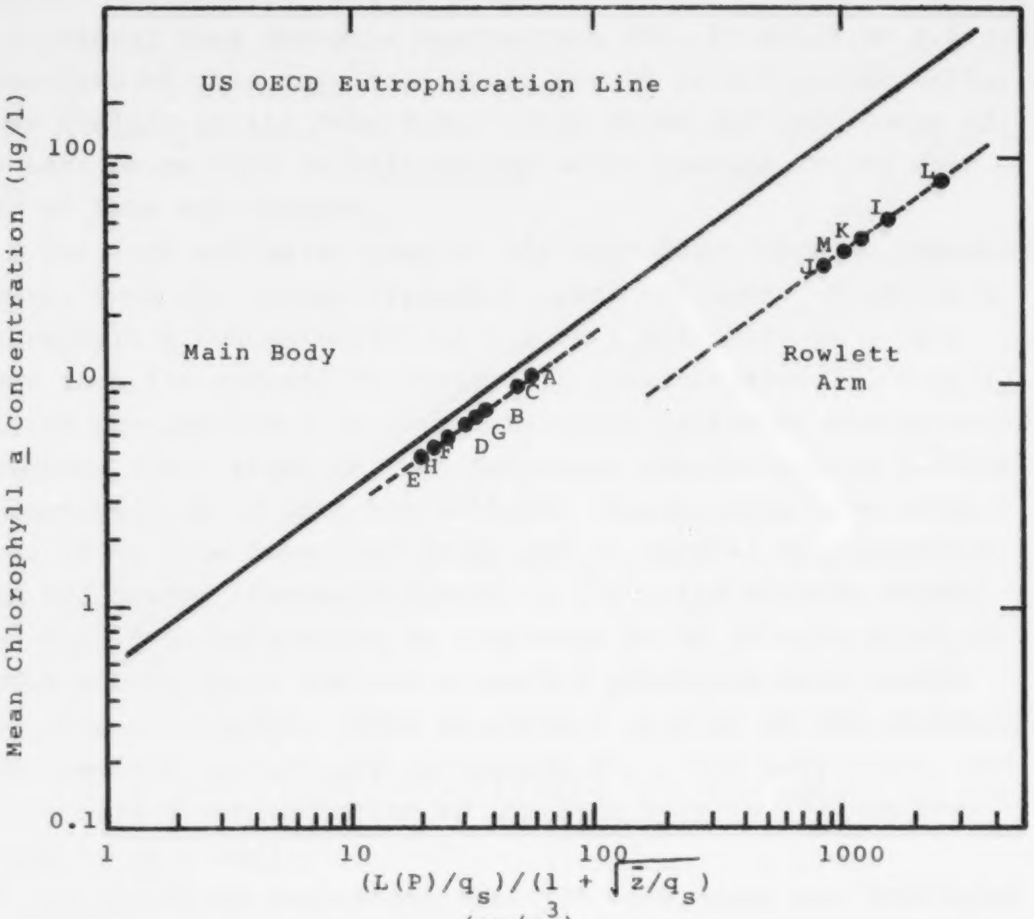

Figure 80. Relationship between phosphorus load and chlorophyll a in the Main Body and Rowlett Arm of Lake Ray Hubbard.

When the Lake Lavon flow and loading are considered with the 1976-77 phosphorus loading (D), the chlorophyll a concentration is reduced by several µg/l. Without any removal of phosphorus from domestic wastewaters, the chlorophyll a concentration is predicted to be 6.5 µg/l. With 85 percent P removal from domestic wastewaters (E), it would be 4.5 µg/l; and with 35 percent removal from domestic wastewaters (F), it would be 5.5 µg/l. Reductions of this magnitude would result in noticeably better water quality in the Main Body. This shows the importance of the Lake Lavon flow in influencing water quality in the Main Body of Lake Ray Hubbard.

The 1990 estimated load to the Main Body, without phosphorus removal from the sewage treatment plant effluents, predicts a chlorophyll a concentration of 7.5 µg/l (G) which is 4 µg/l lower than the current chlorophyll a concentration (11.5 µg/l). This is because the flow from Lake Lavon decreases the hydraulic residence time, which in turn decreases the phosphorus loading characteristics of Lake Ray Hubbard. Under worst-case conditions of no flow from Lake Lavon and no removal of phosphorus from the sewage treatment plants in 1990, the average summer chlorophyll concentration is predicted to be 14 µg/l which is approximately twice the concentration predicted with normal flow from Lake Lavon. With 85 percent removal of the phosphorus from domestic wastewaters and normal flow from Lake Lavon, the chlorophyll a concentration in the Main Body in 1990 is predicted to be 5 µg/l.

It should be emphasized that the OECD model was developed to predict the average summer or growing season epilimnetic chlorophyll a concentration and does not attempt to predict the peak concentrations during the algal blooms of summer, and, in the case of Lake Ray Hubbard, in January.

It is concluded that the water quality of the Main Body, as measured by chlorophyll a concentration, can be significantly improved by removing 85 percent of the phosphorus from domestic wastewaters and by releasing approximately 1.9×10^{11} l/yr from Lake Lavon. It is predicted that by 1990, the average chlorophyll a concentration in the Main Body will increase from 4.5 to 5 µg/l, which would not result in a detectable difference in water quality, if 85 percent removal of the phosphorus from domestic wastewaters was practiced and the flow from Lake Lavon was maintained.

The chlorophyll-phosphorus load relationships for Rowlett Arm, as shown in Figure 80, demonstrate that there is less chlorophyll in this part of the lake than predicted based on the phosphorus loads (I). The model predicts an average summer epilimnetic concentration of about 200 µg/l, However, the average concentration found during the study period was 47 µg/l.

There are two factors which can explain the lower than predicted summer chlorophyll levels in this part of the lake. First, there is evidence that at certain times phosphorus is clearly surplus with respect to limiting algal growth in Rowlett Arm. Second, there is evidence that the high turbidity of this area reduces the rate of photosynthesis and, therefore, the rates of algal growth. Both of these factors should result in lower chlorophyll concentrations than predicted by the phosphorus load to Rowlett Arm, as found in this study.

The removal of 85 percent of the phosphorus from domestic wastewaters entering Rowlett Arm would likely change the summer mean chlorophyll content of this part of the lake by 26 percent to a value of 35 µg/l (J). In order to estimate this change, it is necessary to assume that the phosphorus load-chlorophyll line for Rowlett Arm would be parallel to the OECD eutrophication study line of best fit (i.e., the relationship expressed by the line of best fit also occurs in Lake Ray Hubbard) and that the light limitation that exists now would continue to exist in the future. Since algal growth in Rowlett Arm was nitrogen-limited during some months of the study, it is likely that reducing the phosphorus load by removing 85 percent of the phosphorus in domestic wastewaters would drive this part of the lake to phosphorus limitation during the summer months, which is the crucial time in terms of algal blooms and their associated water quality problems. With summer mean chlorophyll concentrations of 35 μg/l, this area of the lake would still be classified as hypereutrophic. A 35 percent removal (K) of phosphorus from domestic wastewaters would do little to improve the water quality of Rowlett Arm. As shown in Figure 80, the average summer chlorophyll a concentration would be about 40 μg/l.

Based on the chlorophyll concentration predicted from the 1990 estimated phosphorus load to Rowlett Arm, the water quality of this part of the lake will be significantly degraded from its present state. Without any phosphorus removal, the average summer epilimnetic chlorophyll a concentration will be 80 μg/l. However, if 85 percent of the phosphorus is removed from domestic wastewaters, the predicted concentration in 1990 is 35 μg/l, which is 26 percent lower than the current average concentration in Rowlett Arm. This indicates that the water quality in Rowlett Arm in 1990 can be better than it is today if phosphorus is removed by advanced wastewater treatment. However, the high chlorophyll levels which would be present even under the 85 percent P removal strategy would still be considered representative of a hypereutrophic water body, under the Vollenweider (1976) criteria.

Figure 81 shows the relationship between the average summer Secchi depth and the phosphorus loading, normalized by the mean depth and hydraulic residence time, for the US OECD water bodies reviewed by Rast and Lee (1977). As shown in the figure, the Secchi depths of both the Main Body and Rowlett Arm fall below those predicted from the normalized phosphorus loads. The low Secchi depths in Lake Ray Hubbard are likely due to the significant stirring of the sediments into the water column. This reduces the light penetration and the resultant Secchi depth. If

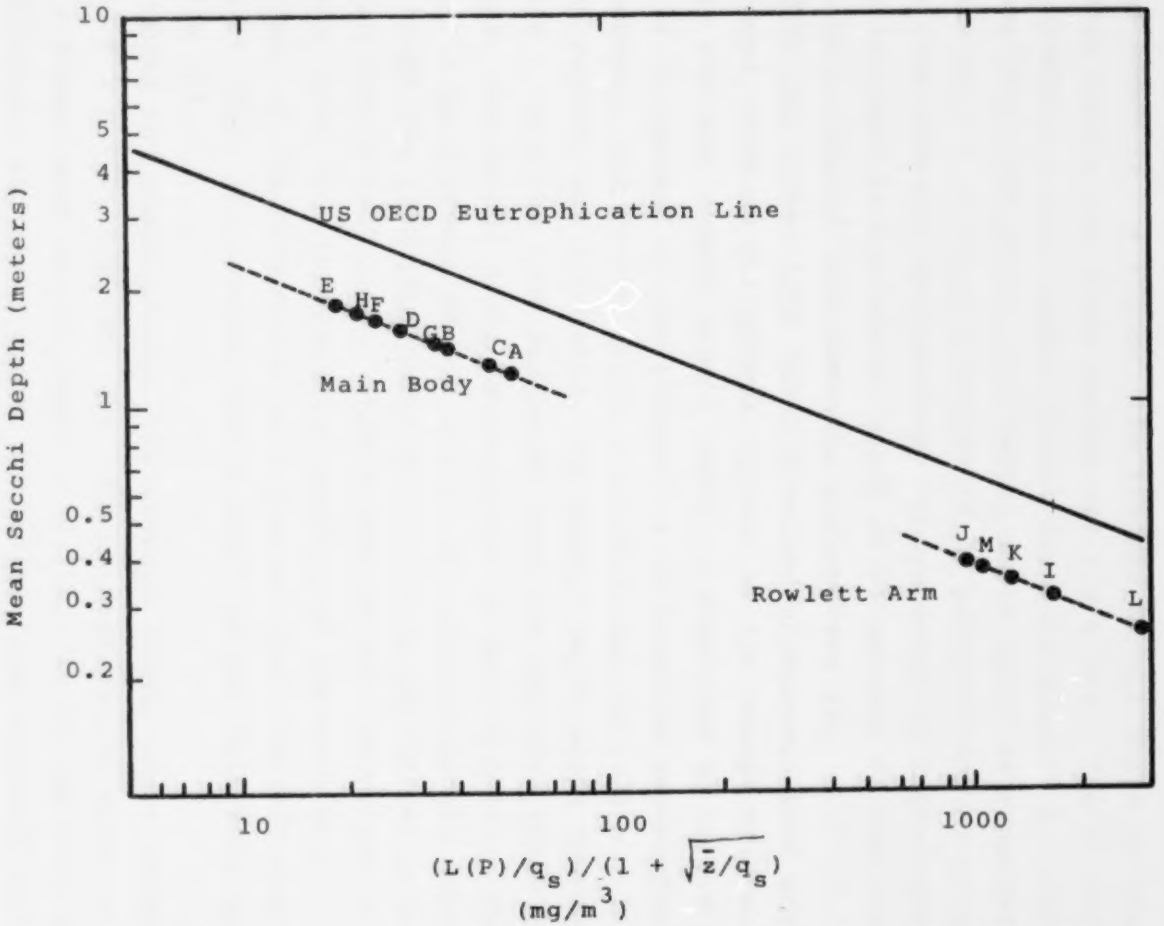

Figure 81. Relationship between phosphorus load and Secchi depth in the Main Body and Rowlett Arm of Lake Ray Hubbard

it is assumed that the turbidity stirred from the bottom will be constant, independent of the chlorophyll content, then it is reasonable to predict the expected changes in Secchi depth assuming a constant proportion between the US OECD eutrophication study line of best fit and the line of existing and predicted values for Secchi depth.

As shown in the figure, the average Secchi depth of the Main Body during the study period was 1.3 m (A). The US OECD model predicts a mean summer Secchi depth of about 2 m. Using the approach described above, the Secchi depth would be expected to increase to 1.4 m (C) as a result of 35 percent removal of phosphorus from domestic wastewaters. There would be little additional increase in the Secchi depth if 85 percent of the phosphorus were removed from domestic wastewaters (B).

When the normal Lake Lavon flow and phosphorus load are considered, with no phosphorus removal at the sewage treatment plants, the mean summer Secchi depth is predicted to be 1.6 m (D). If 35 percent of the phosphorus in domestic wastewaters were removed, the Secchi depth would increase to 1.75 m. If 85 percent removal was initiated, the Secchi depth would increase to 1.9 m. With the 1990 estimated load and the Lake Lavon load and flow, the Secchi depth is predicted to be 1.5 m (G). This represents an increase of 0.2 m over the current Secchi depth, even though the 1990 phosphorus load will be considerably larger than the 1976-77 load. This indicates the importance of the Lake Lavon flow on the water quality of the Main Body. If 85 percent of the phosphorus were removed from domestic wastewaters in 1990, the average Secchi depth of the Main Body would be 1.8 m (H).

During the study period, the average Secchi depth in Rowlett Arm was 0.3 m (I); however, the OECD eutrophication model predicts a summer mean Secchi depth of 0.6 m. As in the Main Body, the difference is likely due to turbidity from suspended sediments. With 85 percent removal of phosphorus from the sewage treatment plants, the Secchi depth would increase to 0.4 m (J).

If 35 percent of the phosphorus were removed from domestic wastewater, there would be no detectable change in the Secchi depth (K). When the 1990 phosphorus loads are considered, the Secchi depth decreases to 0.25 m (L) with no phosphorus removal and increases slightly to 0.35 m (M) if 85 percent of the phosphorus in domestic wastewaters is removed. This shows that reducing the phosphorus load will have a minimal impact on the water clarity of Rowlett Arm.

The long-term average rainfall in the Dallas-Fort Worth region is 5.1 to 7.6 cm per month. However, this area of the country is noted for long periods of drought in which many months may go by with little or no precipitation, and then a 12 cm rainfall will occur in one afternoon. Further, there are also longer-term drought cycles in which periods of years may go by with limited rainfall compared to other years. It is necessary to estimate the potential influence of periods of wet and dry weather on the phosphorus loads which influence algal growth in Lake Ray Hubbard. As noted above, the phosphorus loading which is used to predict the average chlorophyll level, is normalized by the mean depth and hydraulic residence time of the water body. Since periods of drought or extremely wet weather can significantly affect the inflow, which in turn directly affects the hydraulic residence time, it is of interest to calculate the hydraulic residence time and phosphorus loading for the expected ten-year low flow and expected ten-year high flow. The DWU provided estimates of the hydraulic residence times for these flows. The normal hydraulic residence time of Lake Ray Hubbard is about 5.5 years and with normal Lake Lavon flow it is between one and two years. According to the DWU, the hydraulic residence time of Lake Ray Hubbard during the ten-year wet period will decrease to approximately 0.8 years. During the ten-year drought, this would increase to 15.2 years. In order to estimate the impact of these altered flows on the response of the lake to the nutrient load, it was necessary to estimate the changes in nutrient loads as a function of flow. At this time, no definitive information is available on how the nutrient loads to the lake would change as a function of the annual precipitation. It is reasonable to assume, however, that in dry years much less of the land-derived nutrients would be carried to the lake compared to wet years.

For the purposes of the computations presented in this section, it was assumed that the change in the nutrient load between the ten-year low flow and ten-year high flow to the lake would be directly proportional to the flow and that the sewage treatment plants would continue to add nutrients at the same rate as that of the study period. This is likely to be in error because high flow would tend to transport more unavailable forms of nutrients than would be transported under lesser flow conditions. Under low flow conditions, many parts of the watershed yield no land runoff due to the construction of the small impoundments-tanks on the tributary streams. Therefore, the actual magnitude of the changes would likely be somewhat less than predicted.

Using the hydraulic residence times of Lake Ray Hubbard for the ten-year high and low flow provided by DWU, the phosphorus loading characteristics of the Main Body were calculated. Table 59 summarizes the phosphorus loading calculations based on these hydraulic residence times. The flows to the lake were calculated by dividing the volume of the lake by the hydraulic residence times. Using these calculated flows and the volume of the Main Body, the hydraulic residence times of the Main Body were calculated to be 8.8 years during the ten-year low flow conditions and 0.47 years during the ten-year high flow conditions.

The calculation of the phosphorus load to the Main Body was based on several assumptions. First, as discussed above, it was assumed that the nutrient concentration per liter of water flowing into the lake was independent of flow. The non-point source phosphorus loadings under low and high flow conditions were calculated by determining the study period phosphorus loading due to non-point sources, dividing by the 1976-77 flow to determine

TABLE 59. PHOSPHORUS LOADING CALCULATIONS BASED ON TEN-YEAR HIGH AND LOW FLOW-HYDRAULIC RESIDENCE TIMES OF LAKE RAY HUBBARD

	Low Flow	High Flow
τ_ω (Lake Ray Hubbard)	15.19 yrs	0.82 yrs
Volume (Lake Ray Hubbard)	6.04×10^{11} l	6.04×10^{11} l
Inflow	3.98×10^{10} l/yr	7.37×10^{11} l/yr
Volume (Main Body)	3.6×10^{11} l	3.5×10^{11} l
τ_ω (Main Body)	8.8 yrs	0.47 yrs
Point Source P Load (1976-77)	2.38×10^{5} kg/yr	2.38×10^{5} kg/yr
Non-Point Source P Load (1976-77)	1.23×10^{5} kg/yr	1.23×10^{5} kg/yr
Non-Point Source P Load (predicted)	4.92×10^{4} kg/yr	9.07×10^{5} kg/yr
Total P Load-Lake Ray Hubbard (predicted)	1.64×10^{5} kg/yr	1.02×10^{6} kg/yr
Total P Load-Main Body (predicted)	1.19×10^{4} kg/yr	5.47×10^{4} kg/yr
Areal P Load L(P)	0.313 $g/m^2/yr$	1.442 $g/m^2/yr$
$\bar{Z}/\tau_\omega$	1.05 m/yr	19.6 m/yr
$\frac{L(P)/q_s}{1+\sqrt{\bar{Z}/q_s}}$	78.8	39.0

the average concentration, and then multiplying this number by the low and high flows. The phosphorus loading during low flow conditions was calculated to be 1.64×10^5 kg/yr and during high flow conditions it was 1.02×10^6 kg/yr. The second assumption was that 95 percent of the phosphorus would be retained in the arms, independent of flow. This produced phosphorus loads to the Main Body of 1.19×10^4 kg/yr and 5.47×10^4 kg/yr during low and high flow conditions, respectively. Using these phosphorus loads, the areal P load and the P load normalized for mean depth and hydraulic residence time were calculated. These values are presented in Table 59.

Figure 82 shows the phosphorus loading during low and high flow conditions as a function of the hydraulic residence time and mean depth. The 1976-77 loading with Lake Lavon flow and P load-is shown for comparison. It can be seen that the effect of low flow is to place the P loading above the excessive loading line. When the mean depth/hydraulic residence time relationship value is small, a small change in the P load has a large impact on the water quality of the lake. When the Lake Lavon flow is considered along with the 1976-77 loading, the data point drops down to the excessive loading boundary line showing the importance of flow in determining the trophic state of a lake. When the ten-year high flow is considered, the point is significantly above the excessive loading line. The higher flows should have the effect of increasing water quality. However, the increased flows also bring in more nutrients so the total nutrient loading is also increased. It appears that the detrimental effect due to the increased phosphorus loading is greater than the beneficial effect of increased flow, placing the point above the excessive loading line.

Figure 83 shows the effect of flow on the phosphorus loading chlorophyll a relationship. During periods of low flow, the mean summer chlorophyll a is predicted to be approximately 16 μg/l, which is approximately 10 μg/l higher than the average chlorophyll predicted with the 1976-77 loads with the Lake Lavon

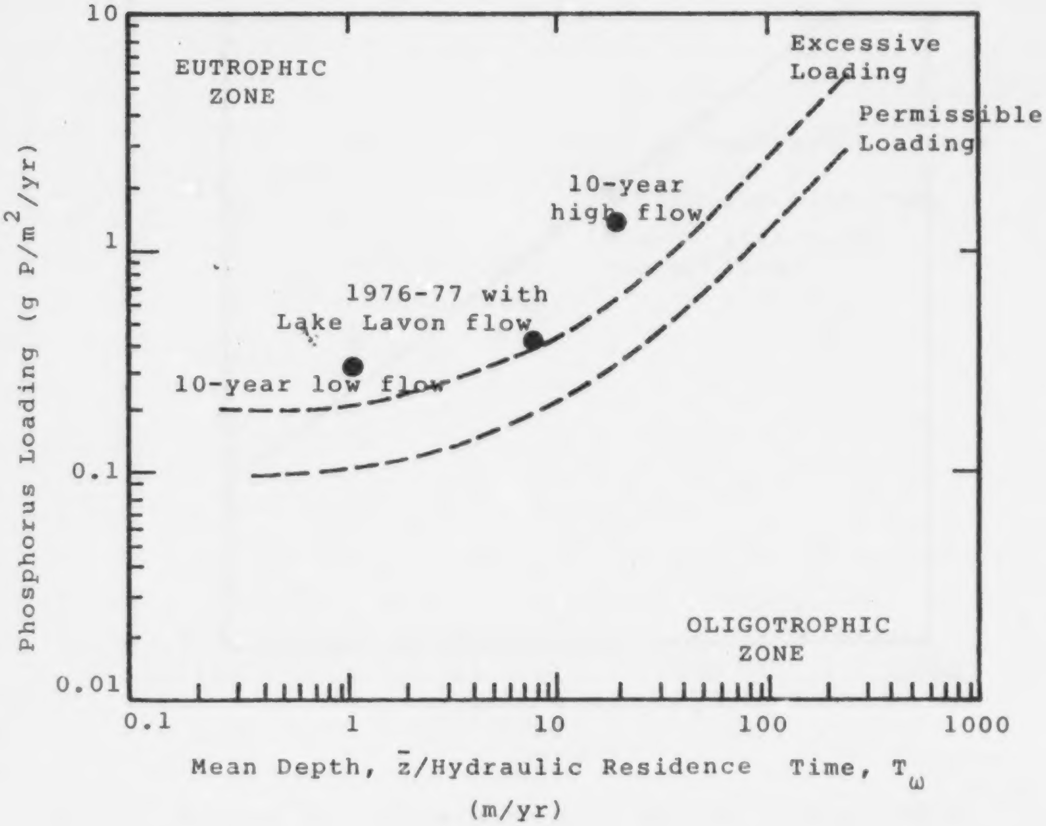

Figure 82. Effect of flow on phosphorus loading to the Main Body of Lake Ray Hubbard

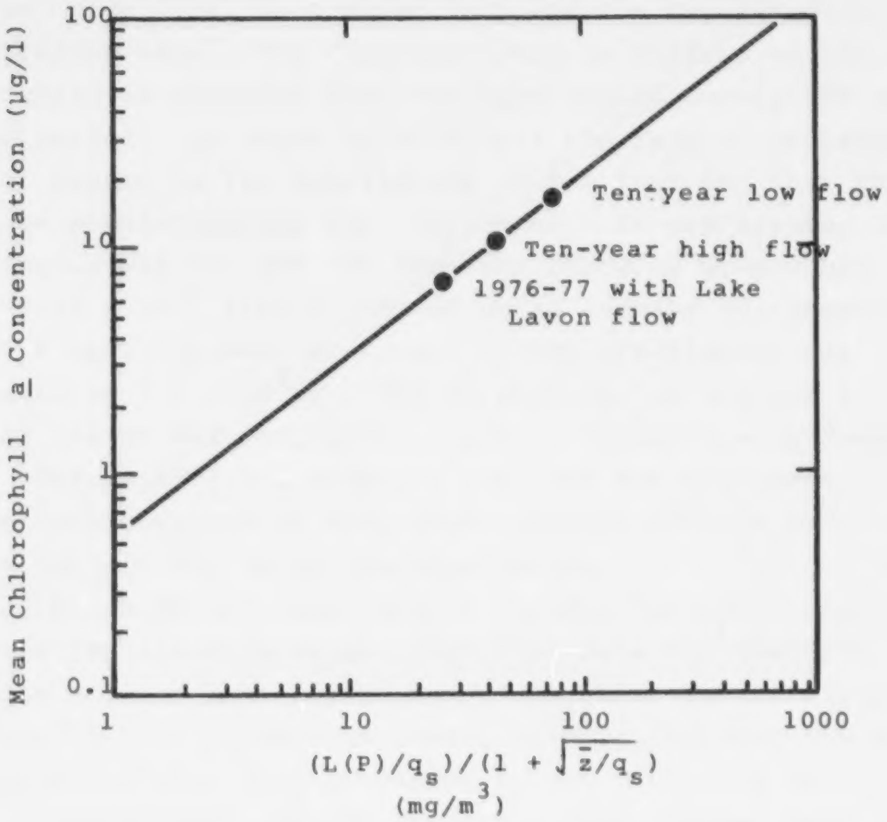

Figure 83. Effect of flow on the relationship between phosphorus load and chlorophyll a in the Main Body of Lake Ray Hubbard

flow. During periods of high flow, the average summer chlorophyll a is predicted to be approximately 10 µg/l. It is evident from these calculations that year-to-year variations in the flow to Lake Ray Hubbard could have a significant effect on water quality within the lake.

Rast and Lee (1977) have shown that there is a relationship between the phosphorus load, normalized for mean depth and hydraulic residence time, to a water body and the hypolimnetic oxygen depletion rate. The depletion rate is defined as the rate at which oxygen is depleted from the hypolimnion during the stratification period. In order to calculate the rate of depletion the mass of oxygen in the hypolimnion on the last day that the water column was isothermal was calculated. It was assumed that the hypolimnion was the area of the lake below 10 m, and had a volume of 1.77×10^{11} liters. Based on an average DO concentration of 6 mg/l the mass of oxygen in the hypolimnion was determined to be 8.8×10^{8} g. The DO profiles at Station 1 showed that the DO was completely depleted below 10 m on June 14, 1977. May 1, 1977 was taken as the last day the water column was isothermal so it took approximately 35 days to completely deplete the DO in the hypolimnion.

Figure 84 shows the relationship between the phosphorus load and the hypolimnetic oxygen depletion rate for the OECD water bodies and for Lake Ray Hubbard. As shown in the figure, based on the 1976-77 phosphorus loads, Lake Ray Hubbard has a greater depletion rate than predicted by the OECD line of best fit. This greater rate of depletion likely stems from the small volume of the hypolimnion of this lake. Oxygen is depleted in the hypolimnion by the decomposition of organic matter (dead algal cells) raining down from the surface waters and by the sediment oxygen demand. The oxygen depletion rate is related to the phosphorus load because the amount of organic matter entering the hypolimnion from the surface waters

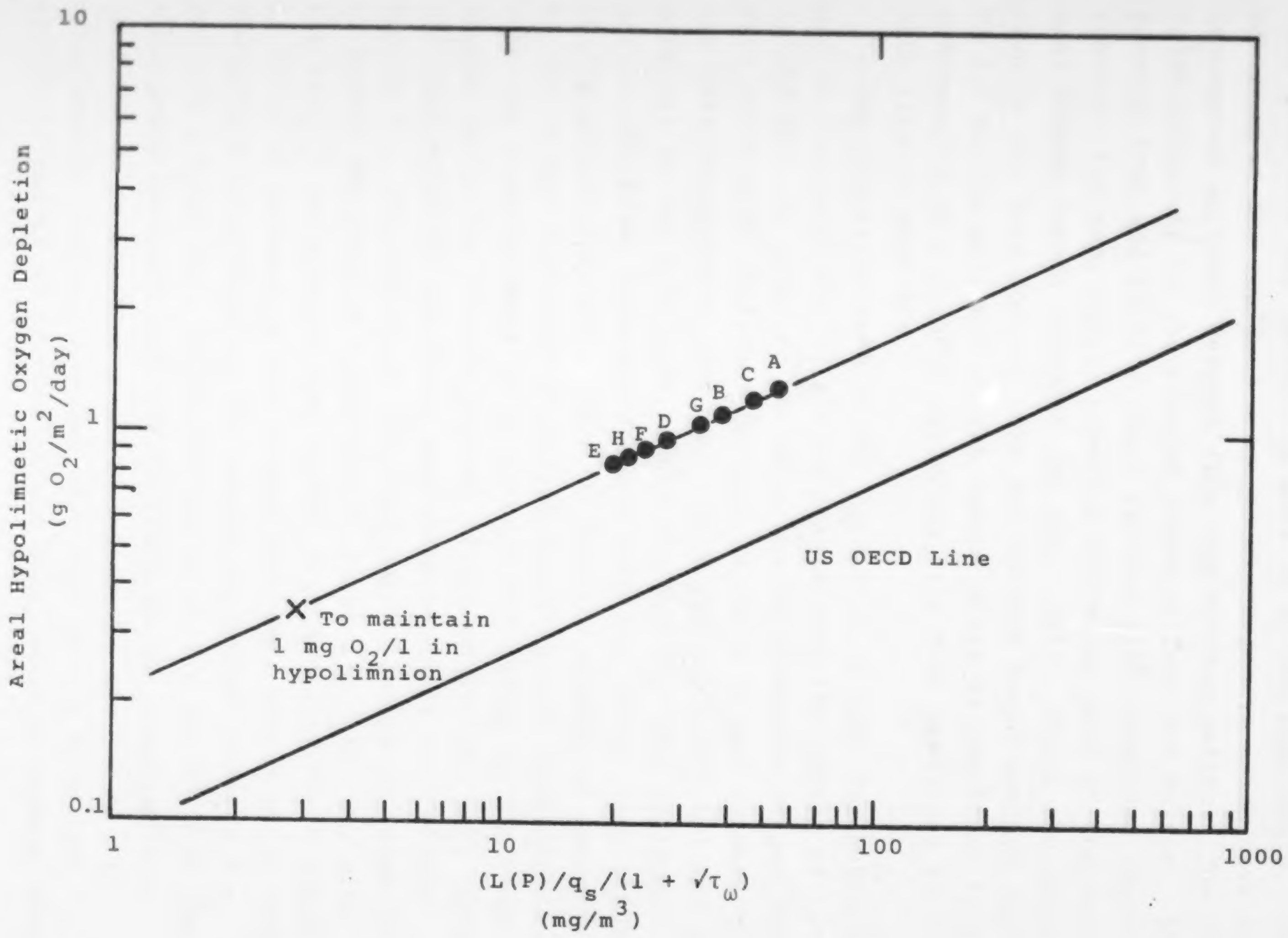

Figure 84. Relationship between phosphorus loading and hypolimnetic oxygen depletion rate

is related to the P load. The sediment oxygen demand of a water body like Lake Ray Hubbard is made up of two components. One of these is any residual decomposible organics which have not decomposed in their descent from the surface waters. The other major component is the reduced forms of iron and sulfur. Both ferrous iron and sulfide react rapidly with dissolved oxygen, thereby, for many lakes, creating the major part of the sediment oxygen demand (Gardner and Lee, 1965). Since the hypolimnion in the Main Body of Lake Ray Hubbard has a maximum depth of 3.5 m, the sediment oxygen demand plays an important role and results in a greater depletion rate than predicted by the OECD line of best fit.

The depletion rate in the Main Body of Lake Ray Hubbard was on the order of 1.2 g O_2/m^2/day during the summer of 1977 (point A). In order to estimate the hypolimnetic oxygen depletion rates with different phosphorus loads it was assumed that the Lake Ray Hubbard line would pass through point (A) and be parallel to the OECD line. This assumes that the sediment oxygen demand of Lake Ray Hubbard does not change with changes in the phosphorus load. With 35 percent removal of phosphorus at the sewage treatment plants discharging into tributaries to Lake Ray Hubbard, there is no perceptible change in the rate of oxygen depletion. If 85 percent of the phosphorus were removed the rate would be decreased approximately 1.15 g O_2/m^2/day (point B). The effect of the Lake Lavon loading and flow is to reduce the rate of depletion to about 1 g O_2/m^2/day with the 1976-77 phosphorus load (point D). Reducing the P loading by 35 or 85 percent at the sewage treatment plants would have the effect of decreasing the oxygen depletion rates to 0.9 and 0.8 g O_2/m^2/day, respectively, (points F and E) when the Lake Lavon load and flow are included in the calculations. In 1990, if no phosphorus control measures are taken, the hypolimnetic oxygen depletion rate will be 1 g O_2/m^2/day (point G) and if 85 percent of the phosphorus is removed from domestic wastewaters the rate will be 0.9 g O_2/m^2/day (point H).

At the present time, the City of Dallas is forced to take epilimnetic water from Lake Ray Hubbard during stratified periods because of the lack of dissolved oxygen in the hypolimnetic waters. As discussed by Lee and Harlin (1965), there may be significant advantages to using hypolimnetic waters since they often contain less algae and have fewer associated taste and odor problems. However, the presence of iron, manganese, and sulfide in the hypolimnetic water of many eutrophic lakes prevents the utilization of these waters for domestic water supply purposes. These chemical species do not form if dissolved oxygen is present in the hypolimnion. It is therefore of interest to determine what P load would be necessary to maintain at least 1 mg O_2/liter throughout the summer stratification period.

In order to have 1 mg O_2/liter in the hypolimnion at the end of the summer, the O_2 depletion rate would have to be 0.32 g O_2/m^2/day throughout the summer. This number is based on a period of stratification of 135 days (May 1 to September 15), an average concentration of 6 mg/l at the beginning of the stratification period, and a hypolimnetic volume of 1.47×10^{11} liters. A depletion rate of 0.32 g O_2/m^2/day is obtained with a phosphorus load to the Main Body of 870 kg P/yr using the 1976-77 hydraulic residence time of 3.5 years. It would be impossible to obtain a load of this amount by simply controlling the P load from the sewage treatment plants since this represents a reduction of 200 percent of the current P load from these sources. It would be necessary to control both the point sources and the diffuse sources of phosphorus.

When the Lake Lavon flow is included in the calculations the hydraulic residence time is decreased to 1.2 years. The decreased hydraulic residence time allows 1 mg O_2/liter in the hypolimnion at the end of summer with a phosphorus load of 1843 kg P/year. In order to obtain a load to the Main Body of

1843 kg P/year, it would be necessary to control the phosphorus load from all sources since a reduction of this magnitude represents a 192 percent reduction in the load from the sewage treatment plants. Therefore, it appears unlikely that the phosphorus load will ever be reduced enough to maintain oxygen in the hypolimnion of Lake Ray Hubbard during the summer period of thermal stratification.

OVERALL EVALUATION OF OECD EUTROPHICATION MODELING RESULTS

It is possible to bring about measurable changes in the P load to Lake Ray Hubbard but it is evident that to convert Lake Ray Hubbard to an oligotrophic lake, with water quality which would not impair any beneficial use, would require a massive phosphorus control effort which would include control of phosphorus from the atmosphere and land runoff. Since this type of control has never been achieved in other areas, it is unlikely that it could be readily achieved in this region. It is interesting to note that even though domestic wastewaters represent a significant part of the total phosphorus load, 85 percent of phosphorus from these wastewaters will cause a relatively small change in overall water quality as measured by the chlorophyll content of the lake.

The chlorophyll or "greenness" of the Main Body of Lake Ray Hubbard is not a significant problem with respect to use of this water body for either recreational or domestic water supply purposes at this time. The most significant water quality problem is the tastes and odors present in the treated water from the DWU East Side Water Treatment Plant. Since it was not possible in this study to characterize the coupling between chlorophyll levels and taste and odor problems in the water supply, it is not possible to predict the impact of altering the chlorophyll levels by a few micrograms per liter on the taste and odor characteristics of the water. Based on work on a variety of water supply impoundments (Rast and Lee, 1977) it is the impression of the authors that changes of this magnitude would not have a significant impact on the water quality of the Main Body of Lake Ray Hubbard.

It is evident, however, that the "greenness" of the Main Body, and especially Rowlett Arm will be increasing as the population of the Lake Ray Hubbard watershed increases, i.e., approximately doubling in the next 13 years (NCTCOG, (1977b). Based on the results of the Lake Ray Hubbard study, it is likely that there will be increased deterioration of Lake Ray Hubbard due to increased phosphorus input from domestic wastewaters. The cities of Plano, Garland-Rowlett, and possibly Rockwall, should investigate the economic feasibility of attaining 80 to 90 percent removal of phosphorus from the domestic wastewater. The advanced waste treatment plants that have been, or will be, installed for wastewater treatment from these cities should be operated to achieve 85 percent removal of phosphorus from domestic wastewaters, unless it places an unreasonable financial burden on these municipalities. Certainly by the early 1980's, removal of 80 to 90 percent of the phosphorus in domestic wastewaters should be routinely achieved from these cities in order to improve water quality in Lake Ray Hubbard. This practice will minimize further deterioration in the Main Body of the lake and lessen, to some yet undefined degree, the taste and odor problems in the treated water from the Dallas Water Utilities' East Side Water Treatment Plant. This practice will have a significant beneficial effect on water quality in the Rowlett Arm of Lake Ray Hubbard.

SECTION 9

DISCUSSION

This section of the report discusses various aspects of the data obtained in this study that have not been discussed in previous sections. Emphasis is placed on providing an overview of various aspects of Lake Ray Hubbard's nutrient load-response relationships compared to those of other water bodies that have been extensively studied. This section also contains a discussion of the management strategies that should be considered for water quality control in Lake Ray Hubbard and suggests an approach that should be used by governmental agencies in this region to develop the information needed to initiate cost effective water quality control programs.

WATER QUALITY CHARACTERISTICS OF THE LAKE

Lake Ray Hubbard is divided into a number of arms by highway causeways and bridges. This study has shown that each of these arms has somewhat different water quality characteristics which, in a general way, are related to the aquatic plant nutrient input to that part of the lake. In this section of the report, water quality refers to the summation of the water quality characteristics such as impairment of recreational use and use of the water for water supply purposes. The relative classification of water quality in various parts of Lake Ray Hubbard presented in this section is based on the experience of the authors.

Previous studies by the Texas Water Quality Board, Duncanville office, have shown that the Rowlett Arm of Lake Ray Hubbard has significantly degraded water quality which appears to be related to the aquatic plant nutrient input from Rowlett

Creek (Personal Communication, J. Pettitt, 1977). The current study has shown that Rowlett Arm would be classified as hypereutrophic in the normal limnological sense, with chlorophyll levels typically near 50 μg/l during the summer months.

Next in the relative degree of decreased water quality is Muddy Creek Arm. The algal numbers and chlorophyll levels were generally substantially higher than those found in the Main Body of the lake. In general, the water quality within this arm is better than that of the Rowlett Arm, but significantly degraded from that of the Main Body of the lake.

The East Fork Arm of Lake Ray Hubbard has water quality which is similar to that of Muddy Creek Arm. At times, substantial algal blooms were found near Station 4 in this part of the lake, but the chlorophyll levels and algal numbers were normally only slightly higher than those found in the Main Body. Discussions of this part of the lake must be conducted with the knowledge that the flow conditions from Lake Lavon were highly atypical during the study period. This indicates that the water quality conditions in the East Fork Arm during the study may not be representative of what would be found under normal flow conditions.

The best overall water quality in Lake Ray Hubbard is found in the Main Body of the lake. However, the water quality is sufficiently degraded so that DWU finds it to their advantage to not use Lake Ray Hubbard as a domestic water supply during late summer due to the taste and odor characteristics of the treated water. The Main Body would be classified as eutrophic based on the data collected during the study period.

The degraded water quality in all parts of Lake Ray Hubbard was found to be directly or indirectly related to the excessive growth of planktonic algae and possibly other organisms whose growth is dependent on the excretory products of planktonic algae. No significant water quality problems found in this lake could be attributed to attached algae or attached or floating macrophytes. Therefore, the development of a water quality management strategy for Lake Ray Hubbard should be based on limiting the excessive growth of planktonic algae.

FACTORS CONTROLLING PLANKTONIC ALGAL GROWTH IN LAKE RAY HUBBARD

This study has, for the first time, determined the relative significance of the dominant factors controlling algal growth in Lake Ray Hubbard. Based on the ratios of available nitrogen to phosphorus during algal blooms, the single most important factor controlling algal numbers in the Main Body of Lake Ray Hubbard is phosphorus, whereas in Rowlett Arm, nitrogen limitation was evident during several months of the study. As demonstrated by the large algal bloom in January, 1977, other factors such as light and temperature do not control algal biomass in the Main Body. The low primary productivity and high levels of turbidity in Rowlett Arm show that light may, at times, be limiting algal numbers in this area of the lake. The relative water quality in the various arms of Lake Ray Hubbard appears to be related in a general way to the areal phosphorus loads (phosphorus load/surface area of the arm).

At this time, insufficient information is available to determine the significance of zooplankton grazing on algal numbers and types in the lake. According to John Pettitt from the Texas Water Quality Board, Duncanville office, the zooplankton present in Lake Ray Hubbard are the small types such as rotifers. These zooplankton cannot efficiently graze the large colonial blue-greens so they possibly rely on detritus as a source of food. It is possible that some part of the detritus is derived from allochthonous sources (outside of the lake).

Significance of Turbidity

The turbidity of Lake Ray Hubbard, in addition to being of potential importance in affecting algal growth within the lake, is also of importance in influencing the public's response to this growth. As discussed by Lee (1977), the higher turbidity present in impoundments has important implications on the relationships between the nutrient loads to a water body and the response of the public to the water quality which results from these loads. In general, the public perceives changes in water clarity as an important distinguishing characteristic of lake water quality. That is, the public would consider a lake or impoundment "polluted" if they observed that the lake had a high water clarity at certain times of the year and at other times, low water clarity due to algal growth. On the other hand, a lake having more or less the same water clarity independent of algal growth would be judged by the public to be less "polluted". This would be especially true if the other water bodies in the region had essentially the same water clarity.

The classification of lakes as oligotrophic or eutrophic has a number of different bases (see Rast and Lee, 1977). One of the more frequently used bases was that developed by Sawyer in a study of approximately 20 southern Wisconsin impoundments (Lee, (1973a). In essence, he found when the nutrient concentrations in the lake reached certain values, the resultant algal growth caused the public to consider the water body "polluted". Sawyer was studying a group of lakes that had relatively low inorganic turbidity; therefore, the more fertile water bodies showed marked seasonal changes in water clarity. Lee (1977) pointed out that a certain nutrient load to a water body which generally has a moderate-to-high level of inorganic turbidity would not have the same adverse effect on the public's use of the water for recreational purposes as this same nutrient load would have on a water body with little or no inorganic turbidity. It is important that pollution control agencies realize that different degrees of public response to a certain nutrient load-chlorophyll level will exist in various parts of the country. The impairment of the public's use of a lake classified as eutrophic, based on nutrient load, concentrations of nutrients, or chlorophyll levels, will likely be different in one area than in another. It is inappropriate to apply water quality standards based on upper Midwest studies to Dallas-Fort Worth regional water bodies. In a Wisconsin lake 10 to 20 μg/l chlorophyll _a_ would be considered adverse to various types of recreational use, but this level of chlorophyll _a_ in a water body in the Dallas-Fort Worth region would have no adverse impact on the same types of recreational uses.

In connection with the implemention of Section 314-A of PL 92-500, requiring every state to classify its water bodies with respect to the degree of fertility, it is important that an understanding of the effect of certain degrees of fertility on use of the water for domestic water supplies and recreational activities be understood. If a relationship between nutrient load and public response is not developed for southern water bodies, it is likely that relationships developed in other parts of the country will be used in the Dallas-Fort Worth region as a basis for classifying water bodies according to their degrees of fertility. This could result in the unnecessary expenditure of large amounts of funds for nutrient control programs in this region which would bring about little overall change in water quality and the public's use of the water bodies in this area.

The chlorophyll concentrations in the Main Body of the lake do not seem to have a significant adverse effect on the public's use of this area for recreation; but this is not true for Rowlett Arm. Rowlett Arm does not have typical blue-green algal scums found in many other water bodies with this degree of fertility, but it does have an aesthetically unpleasing odor which may make it unattractive to the public for recreational purposes. The water is brown-muddy colored with a lot of suspended debris in it. It is rare, however, that the water takes on the "pea-soup" appearance which characterizes highly fertile water bodies in other parts of the country.

Rowlett Arm, the confluence of this arm with Muddy Creek Arm, and the Main Body of the lake represent potentially important areas for understanding nutrient load-algal response-public response relationships.

The high level of background organic and inorganic turbidity in Lake Ray Hubbard has a significant beneficial impact on water quality by limiting the growth of macrophytes and attached algae-periphyton. Lakes in other parts of the country with the same level of fertility as Lake Ray Hubbard frequently have extensive weed beds (macrophytes) and associated attached algae in the shallow waters of the lake (less than 3 m). Studies in various parts of the world have shown that the growth of attached macrophytes is frequently restricted by the absence of light at the bottom. If it were not for the high levels of inorganic turbidity in the lake, it is likely that extensive parts of Lake Ray Hubbard would have weed beds of sufficient density to seriously impair boating, water skiing, fishing, and swimming. It should be noted that while swimming in Lake Ray Hubbard is prohibited by a City of Dallas statute, extensive swimming takes place within the lake. The statute regulations are not currently enforced. A eutrophication-water quality problem that could occur, but is not occurring in Lake Ray Hubbard at this time, is the growth of floating macrophytes such as duckweed and water hyacinth and hydrilla. Extensive problems of this type are being encountered in a downstream reservoir (Lake Livingston). The growth of these plants is causing significant water quality deterioration in this water body. It is unknown why problems of this type have not developed in the Rowlett Arm of Lake Ray Hubbard since it appears that all of the conditions necessary for the growth of floating macrophytes are present in this arm of the lake.

As noted in other sections of this report, the high turbidity levels and low water clarity are likely due to sediment transport in periods of rainfall, runoff, and wind suspension of bottom sediments. It is interesting to note that the pattern of lower water clarity in the Main Body during periods of reduced algal growth was not seen in Rowlett Arm. This further supports the position that the thermal stratification found in the Main Body during the summer months was important in controlling wind-induced suspension of the sediments. In a shallow water body such as Rowlett Arm it is possible that organism-induced mixing could play a significant role in suspension of sediments. Of particular concern would be the role of carp and other rough fish. It is the experience of the authors that carp and catfish can suspend large amounts of sediment into the overlying waters through their rooting and mating action.

The suspension of sediment has two opposing effects on eutrophication of a water body. One of these is to reduce photosynthesis through creating light-limiting conditions. Evidence for this effect has been presented previously in connection with the discussion of photosynthesis measurements. The other effect is through influencing the availability of phosphorus. As discussed by Lee (1970), phosphorus release from the interstitial waters can be inhibited by sediment suspension under anoxic conditions because ferrous iron is also released. Upon contact with oxygen in the water column, the ferrous iron is oxidized to ferric which makes the released phosphorus unavailable through coprecipitation reactions. The opposite effect would be expected from suspension of oxic sediments because phosphorus would be released to the overlying waters. Suspension of these sediments would promote mineralization reactions which would lead to greater phosphate release to the water column. This might be of importance in influencing algal growth in the Main Body of the lake, but it would likely be of minor importance in Rowlett Arm since at times algal growth appears to be nitrogen limited in this part of the lake.

NUTRIENT SOURCES AND CYCLING

Since phosphorus appears to be the element limiting the excessive growth of planktonic algae in the Main Body and nitrogen appears to limit algal growth in Rowlett Arm at times, it is important to understand the sources of nitrogen and phosphorus and the factors controlling the availability of nutrient forms in the lake, in order to develop cost-effective water quality management programs. This section of the report discusses the various factors that influence the sources, transport, and availability of nitrogen and phosphorus in the Lake Ray Hubbard watershed and in the lake.

Sources of Available Forms of Nutrients

A discussion of nutrient load-response relationships for lakes or impoundments must consider the available forms of nutrients in evaluating the nutrient load. It has been found that ammonia and nitrate are generally the available forms of nitrogen that can be readily used. There is evidence that some algae can use only one form of these nutrients. When one form becomes depleted to the point that it is limiting some algal species, it does not necessarily limit the total algal biomass since other types of algae which utilize the other form of nitrogen can readily develop.

Organic nitrogen, the dominant form in natural waters, is generally not available for algal growth and must be converted to simpler forms. Some of the simple organic nitrogen forms such as amino acids appear to be available, but their concentrations in natural waters are usually so small they represent insignificant sources of nitrogen (Gardner and Lee, 1975). In order for the bulk of the organic nitrogen compounds to become available, ammonification reactions must take place which convert the organic nitrogen to ammonia. This is often followed by nitrification reactions which lead to the formation of nitrate. Studies by Cowen, Sirisinha, and Lee (1976) have shown that the reactions converting organic nitrogen to ammonia are relatively slow, requiring weeks to months in many cases.

However, many of the organic forms of nitrogen in natural waters can be converted to available forms if they are suspended in the water column. These investigators estimated that about 80 percent of the organic nitrogen compounds present in the rivers and urban drainage that they studied is readily convertible to available forms in a period of one to two months. This work was done in New York State and Madison, Wisconsin; but it is the opinion of the authors that similar results would be obtained if the same kinds of studies were conducted on organic nitrogen forms present in streams in the Dallas-Fort Worth region.

Ammonia nitrogen tends to sorb on clay minerals and, therefore, become unavailable for algal growth. Usually this reaction is readily reversible, and low concentrations of ammonia in the water would tend to release the ammonium ion, i.e., the sorbed species from the surface of the clays, to the water.

A potentially significant manner in which nitrate is removed from lakes is through denitrification reactions. Studies of Brezonik and Lee (1966; 1968) and others, have shown that nitrate is readily converted to nitrogen gas through denitrification in anoxic systems. The water itself may be oxic, such as in Rowlett Arm, and still have denitrification occurring at the sediment-water interface where the sediments are in a reduced (oxygen-free) system. There would also be denitrification occurring in the hypolimnetic waters of the Main Body of the lake during the summer months when these waters are free of oxygen.

As discussed in a previous section, nitrogen fixation is likely occurring in Lake Ray Hubbard. No measurements were made of this reaction in this study. However, based on the experience of the authors in measuring nitrogen fixation in other water bodies, it likely represents an insignificant source of nitrogen for Lake Ray Hubbard.

It is therefore concluded that the best estimate of the amount of available nitrogen entering Lake Ray Hubbard would be the sum of the nitrate, ammonia, and 80 percent of the organic nitrogen. Something less that this amount would be applicable in deeper lakes. In these lakes it would be possible for the particulate forms of organic nitrogen to be carried to deeper waters where the likelihood of suspension in the water column would be considerably less than in Lake Ray Hubbard. Therefore, less nitrogen would be available for algal growth than the amount in Lake Ray Hubbard.

Soluble orthophosphate is the available form of phosphorus in Lake Ray Hubbard. Other forms such as soluble organic phosphate and particulate phosphate are, in general, not readily available to support algal growth. However, some of the organic forms can be converted to available forms through mineralization reactions, and inorganic particulate forms can be released from the particulate matter, especially under anoxic conditions, and contribute to the available phosphorus.

In Lake Ray Hubbard, and possibly other impoundments in the region, the degree of eutrophication of the water body is such that anoxic conditions are created below the permanent thermocline in the summer, which are then likely to lead to the release of phosphorus. However, in Lake Ray Hubbard the phosphorus content of the hypolimnetic waters is elevated compared to that of the surface, indicating that release of phosphorus is occurring; but it appears that this is of minor importance in supplying nutrients to the overlying waters during the summer months. In some lakes, such as Lake Mendota in Madison, Wisconsin, thermocline migration results in significant transfer of phosphorus from the bottom waters to the surface waters during the summer months. Stauffer and Lee (1973) found that this is an important mechanism for controlling algal blooms in this lake during the summer. However, in the Main Body of Lake Ray Hubbard this mechanism does not appear to operate since the thermocline is essentially permanently fixed at a depth of 10 m throughout the summer.

The condensed phosphates are potential sources of phosphorus present in domestic wastewaters which must be considered in any nutrient control program. These are the pyro and tripolyphosphates that are used with detergent formulations as builders. In the early 1970's, approximately 50 percent of the phosphorus in domestic wastewaters was due to the phosphate builders in detergents. Recent computations by Jones and Lee (submitted for publication) have shown that this value is more typically 35 percent. This indicates there has been a significant reduction in the total amount of phosphorus used in detergents throughout the US during the past half a dozen years. In studies of domestic wastewater treatment plants and in natural water systems, the condensed phosphates have been found to hydrolyze to orthophosphate. The studies by Clesceri and Lee (1965a; 1965b) and Shannon and Lee (1966) have shown that, in general, the condensed phosphates are hydrolyzed at sufficiently rapid rates in natural water systems so that they are readily available to support algal growth.

Cowen and Lee (1976a; 1976b) have found that the best estimate of available phosphorus entering a water body like Lake Ray Hubbard is the soluble orthophosphate plus about 20 percent of the difference between the soluble orthophosphate and the total phosphate. With respect to the Main Body of the lake, these numbers have to be further altered as a result of a combination of the precipitation reactions and the algal growth which occur in each of the arms. As discussed in previous sections, the arms of Lake Ray Hubbard appear to be acting as somewhat independent water bodies in removing between 60 and 95 percent of the total phosphorus entering them.

During periods of high runoff, most of the phosphorus entering the lake are the particulate forms of phosphate which are unavailable for algal assimilation. The actual conditions leading to nutrient transport from one part of the watershed in Parker, Texas, have been observed during the study period by the authors. It was found that some parts of the Lake Ray

Hubbard watershed have contributed few nutrients to the lake during the last several years.

The branch of Rowlett Creek which was observed has been dammed, like many other streams in the watershed, to form what are locally called "tanks" or ponds. These small impoundments have a pronounced effect on water and to some extent, nutrient transport to Lake Ray Hubbard. On this branch of the creek there is a two-acre tank with a maximum depth of about 18 feet. There was a period of about a year and a half prior to February 1977 in which there was no water flow downstream from this tank. This situation arose from the fact that in the summer and fall of 1975, a prolonged drought enabled the tanks to essentially dry up in the region. The spring, summer, and fall rains of 1976, while average or above average, did not create sufficient runoff to fill these tanks with the result that there was no flow in the stream during this period. The tank filled for the first time in a year and a half in February 1977. During the spring of 1977, there were two rainfall events which resulted in the filling of the tank and release of water downstream to Rowlett Creek. From late spring through fall, 1977, there was no further release of water from this tank. This situation occurred even though there were a number of 1 to 1.5 inch per 24 hour rainfall events during this period. It is evident that these small ponds (tanks) are having a significant effect on the amount of water reaching Rowlett Creek from some parts of the watershed. A significant part of the runoff associated with the rains that have occurred during the past two years would have runoff to Lake Ray Hubbard without the numerous tanks that exist on its tributaries.

These tanks are detrimental to Lake Ray Hubbard due to the reduced water flow, but there is a concomitant benefit arising from reduced nutrient input. The watershed of the tank that was observed contributed no nutrients to Lake Ray Hubbard from the summer of 1975 until February 1977, and from May 1977 through the fall of 1977. There was runoff to the streams and tanks from the areas adjacent to the streams so nutrients that were deposited on the land were transported to the streams and became incorporated in the sediments of the streams and tanks. The concentrations of phosphorus in some of this runoff water were measured and found to approach those of domestic wastewaters.

The incorporation of nutrients into the sediments of the steams and tanks prevents large proportions of them from becoming available for algal growth in Lake Ray Hubbard. The binding of phosphate arises from the high sorption capacity of the clays of this area and the fact that this is a calcareous region in which the phosphate can be incorporated on the surfaces of the ubiquitous calcium carbonate particles. The phosphate associated with clay minerals and $CaCO_3$ which is transported to Lake Ray Hubbard during periods of high flow as suspended particulate phosphorus would not likely become available to support algal growth in the lake. It is therefore concluded that the tanks are detrimental to Lake Ray Hubbard by reducing the water input but are beneficial to the lake by reducing the phosphorus input.

Calcium Carbonate Precipitation

One of the potentially important mechanisms controlling the availability of phosphorus in Lake Ray Hubbard is its precipitation as hydroxyapatite $Ca_{10}(PO_4)_6(OH)_2$. This calcium phosphate species has been found to be important in controlling the concentrations of available phosphorus in natural water systems. It appears to most readily form under conditions where there is $CaCO_3$ precipitation. Therefore, waters of at least moderate hardness and alkalinity, such as Lake Ray Hubbard, should be examined for the possibility of hydroxyapatite formation. As discussed by Stumm and Morgan (1970), it appears that hydroxyapatite precipitation takes place on the surface of calcite-$CaCO_3$ crystals. In order to investigate the possibility of the formation of hydroxyapatite in Lake Ray Hubbard in association with $CaCO_3$ precipitation, the data obtained in this study for calcium and carbonate concentrations were corrected for ionic strength and the $CaCO_3$ ion activity product was compared to the $CaCO_3$ solubility product. This is most readily accomplished by comparison between the actual pH of the lake water and the pH at which $CaCO_3$ would be just at saturation (pHs). If the measured pH is equal to the pHs, calcium carbonate should neither dissolve nor precipitate out of solution. If the measured pH is higher than the pHs, calcium carbonate would be expected to precipitate, and if pHs is higher than the measured pH, solid calcium carbonate should dissolve.

The pH of saturation was calculated for Stations 1, 4, and 9 using alkalinity, calcium ion concentration, water temperature, and dissolved solids. Dissolved solids were calculated by multiplying specific conductance by a factor of 0.7. The pH of saturation was determined with the following equation:

$$\text{pHs} = A + B - \log(\text{Ca}^{2+}) - \log(\text{alkalinity})$$

A is a function of water temperature.
B is a function of dissolved solids.

Values for A and B can be found in Standard Methods (APHA *et al*., 1976). The Standard Methods formulae enable the computation of the saturation index. This index is the difference between the pH of saturation and the measured pH. If the index is zero, the water is just at saturation. If the saturation index is positive (pHs > measured pH), then it is supersaturated. A negative saturation index would indicate undersaturation. It should be noted that supersaturated conditions do not necessarily mean that $CaCO_3$ precipitation is occurring. The rates of nucleation of $CaCO_3$ and hydroxyapatite are at times very slow at moderate degrees of supersaturation. There are many factors that influence the rate of precipitation of $CaCO_3$. These have been reviewed by Lee and Delfino (1969). A positive saturation index indicates that $CaCO_3$ precipitation is thermodynamically possible. Based on studies in other waters, it is likely to be occurring in Lake Ray Hubbard.

The measured pH values in the surface waters of Station 1 and throughout the water column at Station 4 were at or near (± 1 pH unit) the pH of saturation throughout the study (See Figures 17 and 50 of the Results Section). During the summer months of 1976 and 1977, the measured pH (8.0 to 8.5) was generally higher than the pHs (7.5 to 8.0), indicating that the system was supersaturated and $CaCO_3$ precipitation could occur. During the winter months, the measured pH (7.5 to 8.0) dropped below the pHs (8.0 to 8.5), indicating that the system was undersaturated with respect to calcium carbonate and any solid calcium carbonate present would be expected to dissolve.

At Station 4, the pHs ranged from 7.5 to 8.0 during both summers, whereas the measured pH values ranged from 8.0 to 8.5. This indicates that the system was supersaturated with respect to $CaCO_3$. The pH of the water decreased slightly and the pHs increased slightly during the winter months. For this period, the system was at saturation with respect to calcium carbonate since the measured pH values were essentially equal to the pHs values.

At Station 9, the measured pH was always at or above the pHs, indicating the water was oversaturated with respect to $CaCO_3$, and precipitation could have occurred throughout the study period. There was no significant change during the winter months. During the summer months, the measured pH values ranged from 8.0 to 9.0, and the pHs values ranged from 7.5 to 8.0. In the winter the measured values ranged from 8.0 to 9.0, and the pHs values ranged from 8.0 to 8.25.

It appears that there is a potential for $CaCO_3$ precipitation in Lake Ray Hubbard. Although not directly studied in this investigation, it appears from the specific conductance, alkalinity, and hardness data in the Main Body of the lake, that precipitation is occurring in the surface waters and dissolution is occurring in the hypolimnetic waters. Therefore, it is likely that the available phosphorus in Lake Ray Hubbard is being reduced by some yet-to-be-defined amount as a result of $CaCO_3$ precipitation. Since precipitation occurs primarily during the summer months, the period of greatest algal numbers, the algal photosynthesis during this time tends to limit algal growth as a result of removal of CO_2 with a concomitant increase in pH, $CaCO_3$ precipitation, and coprecipitation of calcium phosphate species.

There are several mechanisms which tend to promote the precipitation of $CaCO_3$ during the summer months. $CaCO_3$ is less soluble at higher temperatures which promotes $CaCO_3$ precipitation during the summer. Another factor which tends to promote the summertime precipitation of $CaCO_3$ is the decreased solubility of CO_2 at elevated temperatures. According to Lee and Delfino (1969), this is an important cause of elevated pH's in lakes during the summer months. As discussed earlier in this report, the several degrees elevated water temperature during the summer of 1977, compared to the summer of 1976, resulted in a decrease in the CO_2 content of the water by a factor of 2 and a significantly elevated average pH of the water during the latter part of the summer.

Lee and Hoadley (1967) have discussed the potential role of organism activity, such as planktonic algal photosynthesis, in raising the pH of natural water systems resulting in $CaCO_3$ precipitation. As discussed by the authors, this should result in coprecipitation of phosphate. More recently, Otsuki and Wetzel (1972) have studied this phenomenon in a Marl lake in southern Michigan. They found significant phosphate incorporation into the $CaCO_3$ precipitate which was formed in this lake. It is unclear whether the precipitation is a true coprecipitation phenomenon or the formation of hydroxyapatite on the surface of the calcite crystals, as discussed by Stumm and Morgan (1970).

In order to determine if phosphorus was precipitating out of the water column in Rowlett Arm, the degree to which Station 14 water would have to be diluted with Main Body water to obtain water with the Station 8 average total P concentration was calculated for the two summer periods of thermal stratification and the winter isothermal period. The average total P concentrations in the Main Body, Station 8, and Station 14, were calculated for each time period. Using the average concentrations, it was found that during the winter months, water with the Station 14 concentration of 2.18 mg P/l would have to be diluted

35 times with water from the Main Body with an average concentration of 0.039 mg P/l to obtain the Station 8 water with an average total P concentration of 0.098 mg P/l.

Since chloride is a conservative (nonreactive) element, it can be used to determine the degree of mixing and dilution that is actually occurring at Station 8 between the Rowlett Arm water and the Main Body water. Using the chloride data, it was determined that during the isothermal period the water in Rowlett Arm was diluted over 10 times with Main Body water. During the summer of 1976, the Station 14 water had to be diluted 200 times with Main Body water to obtain the Station 8 total P concentration. Rowlett Arm was actually diluted with Main Body water about 25 times, based on the chloride dilution data. During the summer of 1977, Rowlett Arm had to be diluted approximately 76 times with Main Body water based on the phosphorus concentrations; however, the chloride concentrations show that the arm was actually diluted with Main Body water about 11 times.

The fact that the dilution required to obtain the Station 8 total P concentrations was always much greater than the dilution actually occurring, based on chloride data, indicates that some other process must be removing phosphorus from Rowlett Arm. As discussed above, $CaCO_3$ precipitation is a potential removal mechanism for phosphate. Since alkalinity is a measure of the $CO_3^=$ concentration in the water, it can be used to determine if $CaCO_3$ precipitation or dissolution could be occurring. The average alkalinities in the Main Body, Station 8, and Station 14 are shown in Table 60. The alkalinity and, therefore, the $CO_3^=$ species concentration decreased from Station 14 to Station 8, indicating that $CaCO_3$ was likely precipitating out of the water column and phosphate could be coprecipitating with the $CaCO_3$.

Another mechanism for removal of phosphate in Rowlett Arm is its incorporation into algal biomass which, upon the death of the organisms, becomes part of the sediments. The relative significance of $CaCO_3$ mediated phosphate removal versus algal biomass mediated phosphate removal in Rowlett Arm is unknown at

TABLE 60. AVERAGE ALKALINITIES

Depth	Main Body	Station 8	Station 14
6/76 to 9/76			
S	91.3	83.4	126.6
B	107.6	88.6	131.1
9/76 to 5/77			
S	106.3	102.7	153.5
B	110.4	110.2	167.0
6/77 to 8/77			
S	99.8	92.9	118.3
B	110.2	93.1	125.1

S = Surface (1 m)

B = Bottom (0.5 m from sediments)

this time. It is the opinion of the authors that both play important roles in limiting the availability of phosphorus that enters the Main Body of the lake from Rowlett Creek.

RELIABILITY OF NUTRIENT LOAD ESTIMATES

This section of the report presents a discussion of the reliability of the estimates of the nutrient loads to Lake Ray Hubbard. It is important to estimate this reliability in order to be certain that choosing one management strategy versus another is not biased because of unreliable estimates of one or more of the nutrient loads. The reliability of estimates of nutrient loads is based on two factors: the adequacy of estimating the nutrient concentrations based on the sampling and chemical analyses and the estimates of flows. It has been found by several investigators, including the authors of this report, that by far the most critical factor in estimating nutrient loads of streams is the accurate estimate of flow (Lee, 1969). The concentrations in these streams vary to some degree with discharge, but the magnitude of the variation is relatively small compared to the variation in flows. It is for this reason that it is possible to make periodic measurements of concentrations of chemicals but daily estimates of flow are needed in a nutrient load study.

In general, in the Lake Ray Hubbard study, the estimates of the chemical concentrations were adequate to describe reasonably well the chemical characteristics of the tributaries during the summer periods where weekly-to-biweekly sampling was used. There may be some problems with the reliability of the chemical analyses during the winter, when three weeks to one month sampling frequencies were used, and especially during periods of high flows in the spring of 1977.

In interpretation of data of the type generated in this study devoted to nutrient load-response relationships, it is important to consider the sensitivity of the decision-making process for a management strategy relative to the reliability of the measurements that are made in the study. Far too often, individuals concerned with technical studies on lakes spend large amounts of time and effort making very precise measurements when such efforts are not justified due to the fact that the management decision on what can be controlled is relatively insensitive to the reliability of the measurements that were made. It has been the experience of the authors that even very crude nutrient load estimates, based on land use and nutrient export coefficients, are often as valid for formulating nutrient control programs as very precisely measured flows and concentrations from the tributaries. It should be possible, based on the results of the Lake Ray Hubbard study, to develop, with very limited amounts of funding, estimated nutrient sources for each of the water bodies within the NCTCOG area. It should be emphasized that this requires someone fairly knowledgeable in the area of nutrient transport and cycling within lakes and cannot be done by someone with only a cursory knowledge of the environmental chemistry of nutrients. As discussed in another section of this report, attempting to utilize the nutrient loads to the arms of Lake Ray Hubbard gave a very distorted picture of the actual nutrient loads which were affecting the Main Body of the lake.

There are few alternative management strategies which can be adopted for control of excessive fertilization of Lake Ray Hubbard. Considering technical and economic feasibility and based on normal practice in the United States and other countries, the only management strategy that appears to be viable for control of excessive fertilization of Lake Ray Hubbard at this time. is that of the initiation of advance waste treatment practices for phosphorus removal.

An assessment should be made as to whether the estimated phosphorus loads to Lake Ray Hubbard were made with a sufficient degree of reliability to permit an evaluation of the advisability of initiating phosphorus removal at the domestic wastewater treatment plants of Plano, Garland-Rowlett, and Rockwall. In an absolute sense there may be appreciable errors in the estimates of the nutrient loads of certain tributaries, but in the formulation of management strategy, these errors are inconsequential.

IMPACT OF ALTERED FLOW OF LAKE LAVON ON RESULTS OF STUDY

As discussed previously, under average flow conditions, Lake Lavon would be expected to contribute 1.9 x 10^{11} l/yr to Lake Ray Hubbard. Based on the US EPA study (US EPA, 1977a),the amounts of phosphorus transported to Lake Ray Hubbard from Lake Lavon during 1974 represents approximately eight percent of the total tributary P load as measured in 1976 and 1977. Based on the information provided by DWU on the expected average flow to Lake Ray Hubbard from Lake Lavon and using the average concentrations of total P (0.068 mg P/l) found by the US EPA in their 1974 study of Lake Lavon, Lake Lavon would contribute about five percent of the total annual phosphorus load to Lake Ray Hubbard.

It is interesting to note that even though the average expected discharge from Lake Lavon is greater than the total water input from all other tributaries, the amount of phosphorus added to the lake from Lake Lavon is relatively insignificant. According to the US EPA's study, if Lake Lavon were not present in the Lake Ray Hubbard watershed, Lake Ray Hubbard would receive an additional 88,000 kg P/yr. As shown in Table 34 of the nutrient load results section, the total load to Lake Ray Hubbard from June 1976 through May 1977 was 247,000 kg P/yr. The additional 88,000 kg P/yr which currently enter Lake Lavon represent 26 percent of the total annual load which would enter Lake Ray Hubbard if Lake Lavon were not upstream. According to the US EPA report, 20,000 kg P/yr were released from Lake Lavon in 1974. This indicates that 77 percent of the phosphorus entering Lake Lavon was removed in the lake. This type of removal is typical for lakes and points to the significant impact that an upstream lake may have on the transport of nutrients downstream.

The significance of the phosphorus added from Lake Lavon to the Main Body of the lake must be judged in terms of the amount that may be retained in the East Fork Arm of Lake Ray Hubbard. Under normal flow conditions, based on DWU estimates, the hydraulic residence time of water in the East Fork Arm would be on the order of 0.7 years.

Residence times of this magnitude are sufficient to result in large amounts of phosphate removal or incorporation into unavailable forms in shallow water bodies such as the East Fork Arm of Lake Ray Hubbard. This is adequate time to enable algal growth to develop and thereby convert the available forms of the nitrogen and phosphorus entering Lake Ray Hubbard from Lake Lavon into unavailable forms. The East Fork Arm is acting much like Rowlett Arm in this respect. It is evident that the wastewaters from the City of Rockwall which enter near Station 4 in the East Fork Arm are stimulating algal growth in that region. It is also evident, as discussed in other sections, that a substantial part of the Rockwall sewage treatment plant phosphorus does not reach the Main Body of the lake in an available form. Similarly, some of the nutrients entering Lake Lavon would be made unavailable for stimulation of algal growth in the Main Body of Lake Ray Hubbard.

There is another aspect of this situation which must be considered. The increased flow through the East Fork Arm would tend to drive Station 3 water into the Main Body at a greater rate than occurs now. This could cause an increase in the amount of nutrients entering the Main Body. In order to determine if this would increase the nutrient load to the Main Body, it was necessary to compare the relative concentration of nitrogen and phosphorus compounds at Station 3 to those expected from the discharge waters of Lake Lavon minus any losses that would occur between the upper end of the East Fork Arm and Station 3. It would be expected that a substantial amount of the phosphorus added from Lake Lavon would be lost in the upper parts of the East Fork Arm. Therefore, the net effect of the increased flow would be to increase the nutrient load to the Main Body as a result of the increased flow driving the more nutrient-rich waters from Stations 3 and 4 into the Main Body. The Rockwall sewage treatment plant discharge will become an increasingly important source of nutrients for the Main Body of Lake Ray Hubbard as a result of the increased flows in the East Fork Arm due to releases from Lake Lavon. Rockwall is currently the most significant source of nutrients for the East Fork Arm. As the City of Rockwall increases in population, there will be a greater nutrient load added to the East Fork Arm and, as a result of the flows in the East Fork Arm, there will be a greater nutrient load to the Main Body. Any management program for water quality in Lake Ray Hubbard should include careful evaluation of the potential significance of the Rockwall wastewater discharges to Lake Ray Hubbard.

It is important to examine the flow patterns that would normally occur from Lake Lavon. The information provided by DWU on the expected flows from Lake Lavon shows that the majority of the releases would normally be expected during spring and early summer. As a result, the influence of the Lake Lavon flow on transporting nutrients from the East Fork Arm to the Main Body of the lake would be essentially restricted to the spring months. The conditions that were found during the summer months of the study period would probably be applicable to summers with normal flow from Lake Lavon. The net effect of this increased transport of nutrients from the East Fork Arm to the Main Body would be to raise the spring nutrient content to some extent. This might stimulate a somewhat larger algal peak in late May and early June. It probably would have limited impact on algal growth that occurs from mid to late summer which, in terms of water quality, is much more significant.

IMPACT OF DALLAS POWER AND LIGHT'S GENERATING STATION ON WATER QUALITY IN LAKE RAY HUBBARD

Dallas Power and Light (DP&L) electric utility operates an 890 megawatt electric generating station on the western shore of the Main Body of Lake Ray Hubbard. This electric generating station utilizes on the average 2.1×10^9 liters per day of Lake Ray Hubbard water for cooling. This water is heated on the average of 98 °C above the ambient as a result of condenser passage. Questions are often raised about the impact of once-through cooling on stimulating excessive growths of algae in lakes and impoundments. It is generally recognized that higher temperatures tend to promote an increased rate of growth of algae. Further, the higher temperatures tend to promote the growth of blue-green algae which tend to have a more adverse effect on water quality than other types of algae.

No studies are available which can be used to define the impact of the use of once-through cooling by DP&L's electric generating station on Lake Ray Hubbard, but it is likely that this station would have a limited impact. The studies of Lee and Veith (1971) and Lee and Martin (1975) have shown that normal use of lake water for once-through cooling results in limited impact on water quality in lakes if circulation is not restricted. The increased rates of attached algal growth near the point of discharge of some electric generating plants would not be expected in Lake Ray Hubbard. The period of elevated temperature is typically too short to allow an increase in the number of blue-green algae or other types of planktonic algae. It is possible that by heating the water to 98°C the thermal tolerance limit of some aquatic organisms is exceeded. It is doubtful that this is a significant problem in Lake Ray Hubbard due to the relatively small area of the lake that is affected by the discharge.

Probably the most significant impact of the use of once-through cooling on Lake Ray Hubbard water is that of destruction of zooplankton. It has generally been found that many zooplankton are harmed by condenser passage. The magnitude of the kill associated with once-through cooling varies from about 5 to 35 percent of the organisms entrained dependent on plant design.

Another potential problem associated with the use of once-through cooling is due to chlorine used for slime control in the condenser tubes. No information is available at this time on the chlorination practices being used by DP&L at the Lake Ray Hubbard plant. It is doubtful, however, that even with extensive chlorination this practice would have any significant adverse effect on Lake Ray Hubbard water quality.

It is concluded that DP&L's use of Lake Ray Hubbard water for cooling is not likely to have a significant adverse effect on water quality in Lake Ray Hubbard due to the small size of the generating station, its intermittent operation, and the large volume of Lake Ray Hubbard water available for heat and other potential impact dissipation.

ESTIMATED FUTURE NUTRIENT LOADS TO LAKE RAY HUBBARD

The estimated future nutrient loads to Lake Ray Hubbard should be investigated in a study of this type. The Lake Ray Hubbard watershed is a rapidly growing metropolitan area. Plano, which a few years ago had a population of a few thousand people and today has in excess of 50,000 people, is growing at a rate of approximately 5000 people per year. Garland, the other major metropolitan center in the Lake Ray Hubbard watershed, appears to be growing at a similar rate. Since all of Plano's wastewaters, and a substantial amount of Garland's wastewaters are discharged to Rowlett Creek, it is important to consider the impact this growth in population might have on future nutrient loads to Lake Ray Hubbard. In order to make an estimate of future nutrient loads, it was necessary to make a number of assumptions about the rates of growth of the urban areas in the Lake Ray Hubbard watershed.

In order to estimate the increased P load to Lake Ray Hubbard from the Plano, Garland-Rowlett, Rockwall, and Wylie sewage treatment plants, it was necessary to estimate the expected 1990 populations of the cities based on the expected sewage treatment plant flows provided by NCTCOG (1977a). The 1975 flows were divided by the 1975 populations to obtain the average flow per capita per day for each sewage treatment plant. As shown in Table 61 , these values ranged from 251 l/capita/day from Plano to 404 l/capita/day from Rockwall. The NCTCOG has provided an estimate of the domestic wastewater treatment plant flows for the year 1990. They use three different projections for each of the municipalities of the area. The "B" projections were used for these computations. These assume that growth will continue at the same rate as it has in the past, determined to a major degree by the rate of sewering undeveloped areas. The projected 1990 B sewage treatment plant flows were divided by the 1975 per capita flows to obtain the estimated 1990 populations that would be served by each plant. The projected populations are shown in Table 62. Examination of Tables 61 and 62 shows that the populations of Plano, Rockwall, and Garland will double from 1975 to 1990. It is expected that the sewered population of Wylie will remain the same as it is today. It should be noted that the per capita output of phosphorus from Rockwall is different from that used in other sections of this report. This change reflects the fact that it is expected that considerably improved treatment should be achieved at the Rockwall sewage treatment plant in the future. This approach assumes that the per capita use of water will not change from 1975 to 1990 in each of the municipalities considered. The 1990 phosphorus load from each plant was estimated based on these projected populations and an average contribution of 2.5 lbs P/capita/yr (1.134 kg P/capita/yr). This is a value used by the US EPA in connection with their 1974 National Eutrophication Survey study of lakes and impoundments in the

TABLE 61. POPULATION AND FLOW DATA FOR SEWAGE TREATMENT PLANTS IN THE LAKE RAY HUBBARD WATERSHED

Sewage Treatment Plant	1975 Population Served	Flow ($\ell \times 10^{-6}$)	Flow (ℓ/capita/day)
Plano	51,800	13	251
Garland	34,500	12	348
Rockwall	4,700	1.9	404
Wylie	3,100	0.8	258

Developed from: NCTCOG, Analysis of Sewage Load and Flow Factors (1977).

TABLE 62. ESTIMATED 1990 POPULATIONS AND PHOSPHORUS LOADS FOR THE SEWAGE TREATMENT PLANTS IN THE LAKE RAY HUBBARD WATERSHED

Sewage Treatment Plant	Estimated 1990 Population Served	Estimated 1990 Phosphorus Load* (kg P/yr)	1976-1977 Phosphorus Load (kg P/yr)
Plano	112,000	127,000	66,700
Garland	74,200	84,150	45,535
Rockwall	9,400	10,660	8,868**
Wylie	3,100	3,500	3,500
TOTAL		225,310	114,603

*Based on 2.5 lbs P/capita/yr (1.134 kg P/capita/yr).

**Based on 2.04 kg P/capita/yr - Improved treatment plant efficiency changes per capita P contribution from 2.04 kg/yr to 1.134 kg/yr.

north central Texas region. It is in accord with the value estimated by Jones and Lee (submitted for publication) of 1.4 kg/capita/yr of phosphorus in the influent to domestic wastewater treatment plants. Examination of the per capita phosphorus output from the Garland-Rowlett plant shows that an average of 1.3 kg/capita/yr was discharged in the effluent from this plant. During the study, the Garland-Rowlett plant was badly overloaded. It is felt that with the operation of the new treatment plant currently under construction, the amount of phosphorus discharged should be decreased to some extent. It was felt that it was reasonable to use a value of 1.1 kg/capita/yr discharged from the sewage treatment plants in the Lake Ray Hubbard watershed.

The estimated P loads and the P loads measured during the study are presented in Table 62. The estimated P load of 225,310 kg P/yr for 1990 represents an increase in the P contribution from sewage treatment plants of approximately 110,700 kg P/yr which is an increase of about 97 percent over the current sewage treatment plant load of 114,603 kg P/yr.

In terms of the total annual P load to Lake Ray Hubbard (land runoff, sewage treatment plants, and the atmosphere), the estimated increase in the P load from the sewage treatment plants in 1990 represents an increase of 45 percent of the current P load. The percent of the total P load to Lake Ray Hubbard derived from the domestic wastewater treatment plants will increase from 46 percent in 1976-1977 to 63 percent in 1990.

As indicated previously, the current total P load to Lake Ray Hubbard could be reduced from 247,000 kg P/yr to 150,000 kg P/yr with initiation of 85 percent removal of P from domestic wastewaters. If it is assumed that all domestic wastewaters discharging to Lake Ray Hubbard are treated for P removal achieving an 85 percent P reduction in the year 1990, the expected P load to Lake Ray Hubbard would be approximately 166,000 kg P/yr. Therefore, with 85 percent removal of domestic wastewater phosphorus the 1990 loads, even with the projected increase in population, would be 67 percent of the current load. It is obvious that to the degree that water quality problems in Lake Ray Hubbard are dependent on phosphorus load, there will be a significant reduction in water quality in Lake Ray Hubbard unless steps are taken to remove a substantial part of the domestic wastewater phosphorus entering the lake. If 85 percent phosphorus removal from domestic wastewaters is initiated, the water quality in Lake Ray Hubbard should be considerably better than it is today. A quantitative estimate of the potential changes in water quality associated with projected future phosphorus loads is presented in the section of this report devoted to the OECD modeling.

RATES OF RECOVERY

In any lake, the rate of algal biomass adjustment to a new nutrient load is primarily dependent on the chemical residence time of the species controlling algal growth in the water body. In most of Lake Ray Hubbard, except Rowlett Arm in some months, the key element is phosphorus and it is the phosphorus residence time that must be estimated. Vollenweider (see Rast and Lee, 1977) has shown that the hydraulic residence time of a lake is one of the most important factors governing the lake's primary productivity. The hydraulic residence time is equivalent to the filling time of the lake. Table 63 presents the estimated hydraulic residence times for various parts of Lake Ray Hubbard based on the flows that occurred in 1976-77. Values are also presented in this table for the hydraulic residence times of the East Fork Arm and Main Body, assuming the DWU estimated average flow from Lake Lavon of 1.9×10^{11} l/yr. A review of this table shows that during the study period the Main Body of the lake had a hydraulic residence time of approximately 3.5 years. Under normal flow conditions from Lake Lavon this would be cut to 1.2 years. During the study period, the hydraulic residence time of the East Fork Arm was approximately 8.8 years. When the flow from Lake Lavon is included, the residence time is reduced to 0.7 years. Rowlett Arm was found

TABLE 63. HYDRAULIC AND PHOSPHORUS RESIDENCE TIMES FOR LAKE RAY HUBBARD

Area of Lake	Volume (1)	Inflow (1/yr)	τ_ω/yr	Avg P conc (kg/1)	P loading (kg/yr)	τ_ρ (months)	R%
Main Body	3.5×10^{11}	1.0×10^{11}	3.5	0.04×10^{-6}	1.19×10^{4}	10.7	65
		2.9×10^{11}*	1.2*				52
Rowlett Arm	1.9×10^{10}	7×10^{10}	0.3	0.2×10^{-6}	1.82×10^{5}	0.3	34
East Fork Arm	1.5×10^{11}	1.7×10^{10}	8.8	0.6×10^{-6}	3.41×10^{4}	3.2	75
		2.1×10^{11}*	0.7*		5.41×10^{4}*	2*	46
Muddy Creek Arm	3.0×10^{10}	1.9×10^{10}	1.6	0.1×10^{-6}	2.7×10^{4}	1.3	56

τ_ω = Hydraulic Residence Time
τ_ρ = Phosphorus Residence Time
R% = Phosphorus Retention Time

*Considering Lake Lavon flow (1.9×10^{11} 1/yr) and P load (20,000 kg P/yr) to have a hydraulic residence time of 0.3 years during the study period. Muddy Creek had a hydraulic residence time of 1.6 years.

The hydraulic residence times of each part of the lake are sufficient to allow the algae to utilize any available nutrients entering from the tributaries before the water leaves that part of the lake. This is of importance in that it means the Vollenweider-OECD eutrophication approach for establishing nutrient load-lake response relationships should be applicable to Lake Ray Hubbard and each of the arms of the lake. It should be noted that the hydraulic residence times presented in Table 63 are based strictly on inflow volume and do not consider any exchange of water between the Main Body and the arms. While the overall hydraulic residence time of the lake would remain the same, the residence times of the arms would be altered to some extent based on this exchange.

Table 63 also presents the phosphorus residence times for the various parts of Lake Ray Hubbard. The phosphorus residence time is equivalent to the time necessary under normal flow conditions to achieve the average phosphorus concentration of the lake, assuming the lake started out with no phosphorus in the water column. It is somewhat related to the time that phosphorus is available in the water column to support algal growth. The Main Body of the lake has a residence time of approximately 10.7 months. The phosphorus load to the Main Body was computed by using the concentrations of Stations 3, 8, and 11 and the flows at Stations 15 and 24 on Rowlett Creek, Station 16 on Muddy Creek, and Station 21 on Squabble Creek. The 0.3 month phosphorus residence time of Rowlett Arm reflects the fact that phosphorus is rapidly removed from the water column by algal uptake and precipitation reactions. The 1.3 and 3.2 month phosphorus residence times of Muddy Creek Arm and the East Fork Arm are greater than the Rowlett Arm phosphorus residence time but are still short compared to other water bodies as discussed by Sonzogni et al. (1976).

Water bodies with hydraulic residence times of one to several years tend to have phosphorus residence times on the order of a year. The shorter phosphorus residence times found in Lake Ray Hubbard and its various arms strongly support the observations mentioned in other parts of this report, that a significant amount of the phosphorus added to this lake is in the particulate form which does not become available for algal growth. In studies by Rast and Lee (1977) it has been found that there is an inverse relationship between the algal productivity of a water body, i.e., degree of eutrophication, and the phosphorus residence time. Most productive waters tend to have the shortest phosphorus residence times. This is to be expected since one of the most important mechanisms for removing phosphorus from the water column is its uptake by algae and subsequent death and sedimentation-incorporation into the sediments.

Table 63 includes data on the phosphorus retention coefficient for the various parts of Lake Ray Hubbard. Vollenweider (see Rast and Lee, 1977) has developed this coefficient in order to estimate the percent of the total phosphorus added to a water body which is retained by incorporation into the sediments. It is estimated by the following equation:

$$R = \frac{1}{1 + \rho\omega} \times 100\%$$

where R = phosphorus retention coefficient

$$\rho\omega = \frac{1}{\text{hydraulic residence time(yrs)}}$$

The derivation of this relationship has been reviewed by Rast and Lee (1977). This relationship is equivalent to making a mass balance of inflow and outflow phosphorus, assuming that the phosphorus concentration on an annual basis in the water column is at steady state. Examination of Table 63 shows that from 34 to 75 percent of the phosphorus added to the arms of the lake is retained in the arms. These values are typical of those found in many other lakes. They support the conclusions of other parts of this study, that a large amount of the phosphorus added to the arms of the lake does not reach the Main Body in a form available for algal growth.

One of the primary purposes of computation of phosphorus and hydraulic residence times of Lake Ray Hubbard is to estimate the time the lake would take to adjust to a new phosphorus load. As discussed by Sonzogni et al. (1976), lakes tend to adjust to new load conditions in accord with a first order rate process. This means that about three times the residence time for the chemical species is required to achieve the new equilibrium conditions. For conservative chemicals like chloride which are essentially nonreactive in natural water systems, the chemical residence time equals the hydraulic residence time. This means that a new chloride load for the Main Body of Lake Ray Hubbard would require 3.6 years (3 x 1.2) to achieve a new equilibrium chloride concentration in this part of the lake. These calculations are based on the normal flow from Lake Lavon. For a highly reactive chemical such as phosphorus, with a phosphorus residence time of 10.7 months, 2.7 years would be required to reach a new equilibrium phosphorus level in the Main Body of the lake. This value is of importance in terms of remedial control programs since it means that approximately 2.7 years must pass before the full impact of the control program would be manifested in the water quality of the lake. For the other parts of the lake, where the phosphorus residence times are somewhat shorter, the lake should adjust very rapidly to new phosphorus loads.

It is sometimes indicated in the water pollution literature that highly eutrophic lakes would tend to recover much more slowly from some remedial phosphorus control programs because of release of phosphorus from sediments. At one time it was generally believed by many that phosphorus control programs would likely prove to be ineffective in correcting excessive fertilization problems due to phosphorus. A number of studies have shown that this is not the case. The classic case of reverse eutrophication is that of Lake Washington near Seattle (Edmondson, 1972). Sonzogni et al. (1976) have discussed the impact of diversion of wastewaters from the lower Madison, Wisconsin lakes. Lake Ray Hubbard is more similar to these lakes than Lake Washington, in that they are shallow, essentially nonstratified, and highly eutrophic. The lower Madison lakes also have relatively short hydraulic residence times of a few months. The diversion of Madison's wastewaters around these lakes showed that they adjusted to the new phosphorus load in accord with the model developed by Sonzogni et al. (1976). There was not a large release of phosphorus from the sediments to the water column to slow down the recovery of these lakes. Similar rates of recovery, in accord with the above discussion, would be expected for Lake Ray Hubbard.

One of the major issues that must be resolved in connection with the excessive fertilization of Lake Ray Hubbard is that of the potential significance of the increased algal growth in each of the arms of the lake on the quality of water received at the Dallas water supply intake near the dam. It is possible that the excessive algal growths in the arms of Lake Ray Hubbard could lead to higher levels of tastes and odors than would be present were it not for the algal growth in the arms. It should be noted, however, that the various arms of the lake are, without question, a very significant factor in reducing the algal growth in the Main Body. If the nutrient loads that are now received by the arms were added directly to the Main Body of the lake, there would be a tremendously different water quality within the Main Body. The greater overall clarity of the water in this region, coupled with the increased nutrient input, would result in a much greater algal production than is found today in this region. The adverse effects of algal growth within the arms of Lake Ray Hubbard cannot be quantified at this time, but they are certainly outweighed by the beneficial effects in terms of increased nutrient (phosphorus) removal that takes place within the arms.

Phosphorus retention in the streams leading to Lake Ray Hubbard is another area in which there is interest. The Plano sewage treatment plant's discharge of phosphorus to Rowlett Creek is likely to have a significantly reduced impact on Lake Ray Hubbard than an equivalent amount of discharge from the Garland-Rowlett plant. This relates to the fact that in the streams, phosphorus is converted to forms which are generally not available for algal growth. It is becoming increasingly recognized today that a municipality discharging to a tributary at some distance from a lake has significantly less impact than those discharging directly to the lake. Current work in the Great Lakes basin by Chapra (Personal communication, 1977) shows that under some circumstances, the amount of phosphorus reaching the Great Lakes from municipalities discharging wastewaters to rivers tens of miles from the lake is greatly reduced. The data collected in the Lake Ray Hubbard study show that the amount of available phosphorus from the Plano sewage treatment plant is reduced in Rowlett Creek. This should not be interpreted to mean that the discharge of wastewaters from the Plano sewage treatment plant is not having an adverse effect on water quality in Lake Ray Hubbard. It is certainly having an effect since it is one of the principal sources of phosphorus for the lake. However, its impact per unit of phosphorus discharged to Rowlett Creek would be some unknown amount less than that of Garland-Rowlett. The reduction in available phosphorus in Rowlett Creek from the Plano sewage treatment plant to the point where Rowlett Creek flows into Lake Ray Hubbard would not be sufficiently high to justify not initiating phosphorus removal at the Plano sewage treatment plant. In order to quantify this relationship, a long-term, fairly expensive study would be necessary which would most likely conclude that Plano should initiate phosphorus removal from its domestic wastewaters.

OVERALL CHARACTERISTICS OF PHOSPHORUS IN LAKE RAY HUBBARD

In previous sections of this report the various types of reactions that could influence the behavior of phosphorus in Lake Ray Hubbard have been discussed. In many instances, it was not possible to quantitatively determine the relative significance of each of these factors. In order to gain some potential insight into the significance of each factor, an approach developed by Lee *et al*. (1977) involving examination of the total mass of a nutrient in a lake or part of a lake during the study period should be used. This approach works best when there are data collected over several years on a water body; however, it can be applied to Lake Ray Hubbard.

Figures 77 and 78 are plots of the total P and soluble ortho P masses in the water column and epilimnion of the Main Body during the study period. The difference between the water column mass and the epilimnion mass is the mass of phosphorus in the hypolimnion, which is only present during periods of thermal stratification. The phosphorus masses as shown in Figures 77 and 78 are the result of all factors and reactions which control phosphorus in the lake. Factors such as tributary input of P, recycling from the sediments, precipitation to the sediments, and other removal processes all act to control the total mass of phosphorus in the lake. Changes in the total mass reflect the relative significance of these processes during various times of the year.

Examination of Figure 77 shows that in general, the total mass of phosphorus in the Main Body of Lake Ray Hubbard varied between 1×10^4 and 2.5×10^4 kg. The amount of total P in the epilimnion was approximately 60 to 80 percent of the total mass in the Main Body. There are two major peaks in the total P curves in June 1976 and in March 1977. The cause of the June 1976 peak is unknown at this time. The March 1977 peak was most likely due to the input of phosphorus form the tributaries,

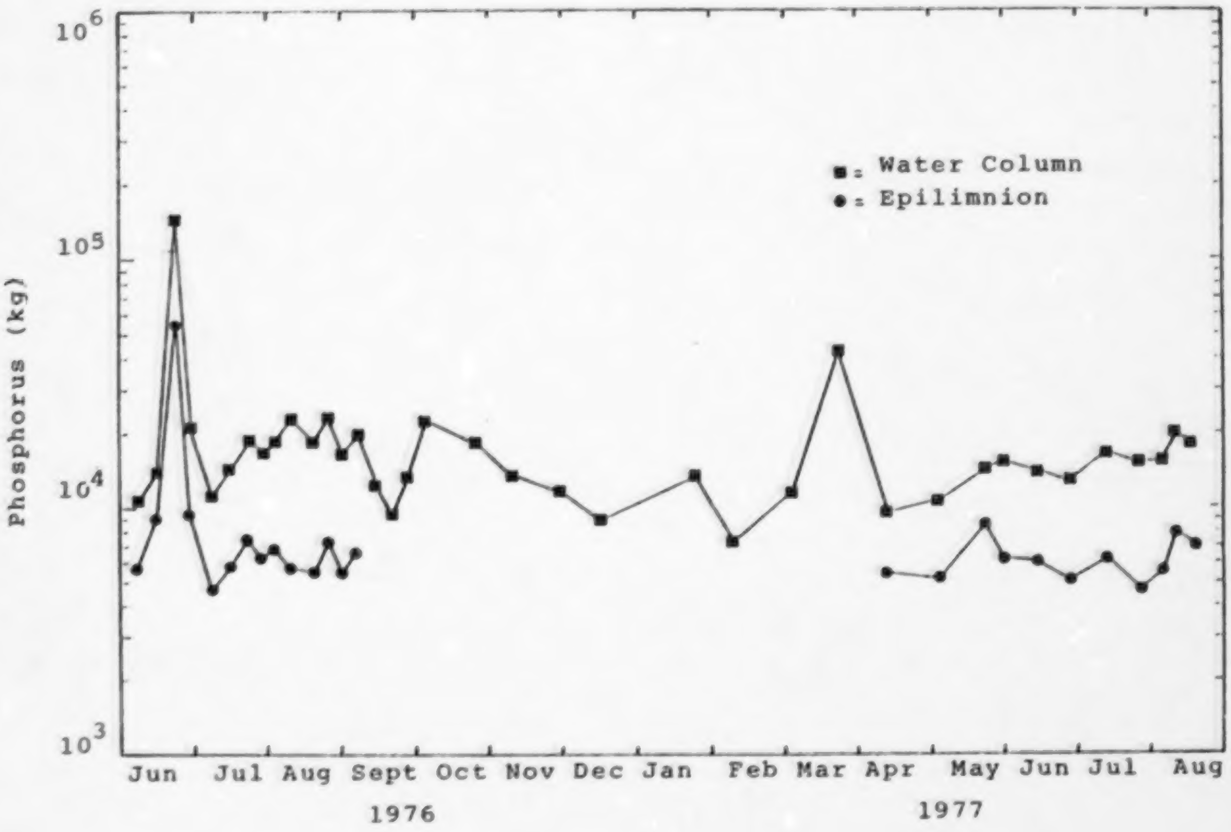

Figure 85. Total phosphorus mass - Main Body

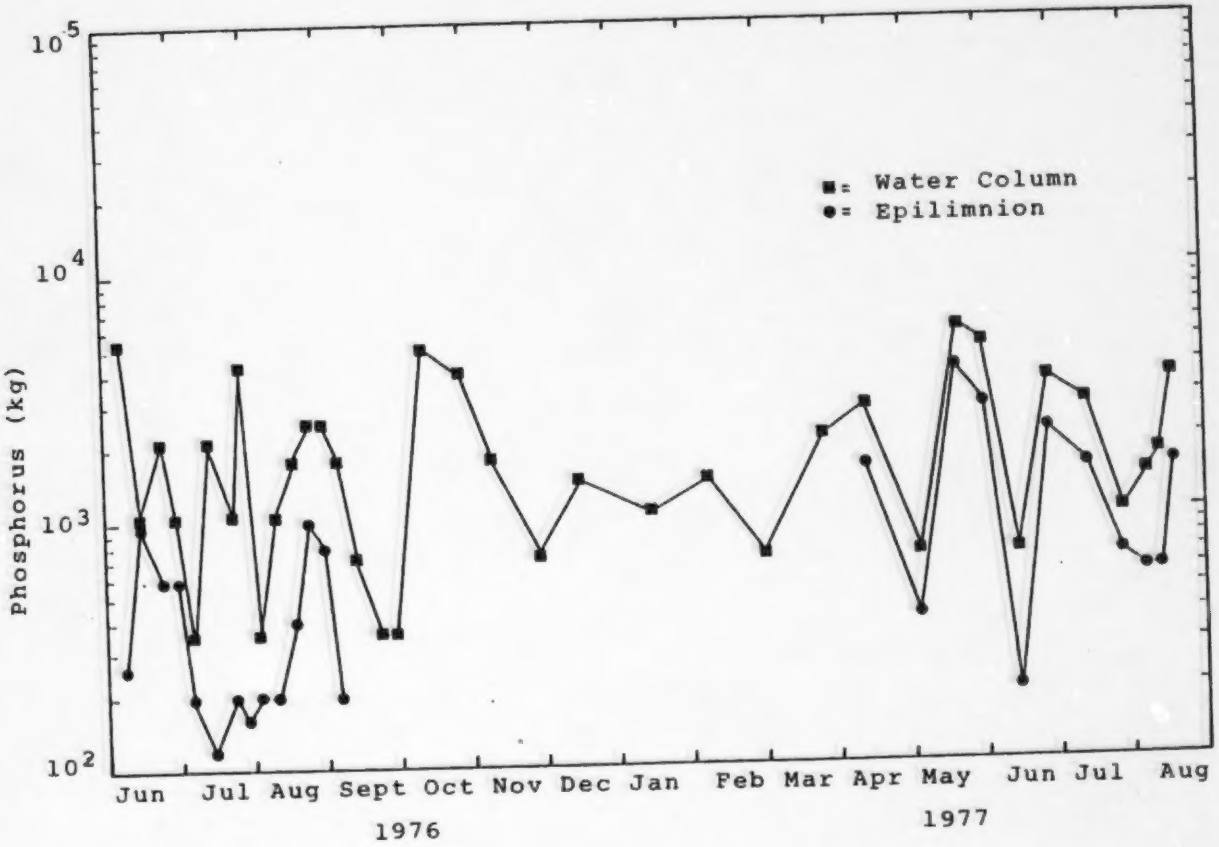

Figure 86. Soluble orthophosphate mass - Main Body

related to the heavy rainfall that occurred during the spring months. As shown in Figure 78 this rainfall also contributed to an increase in the soluble ortho P mass in the Main Body during March. The elevated soluble ortho P mass did not persist and stimulate algal growth during the summer as shown by the decrease in the P mass that occurred in late April 1977. There was a decrease in the total P and soluble ortho P masses at fall overturn in September 1976. It is interesting to note that there was an increase in the total P and soluble ortho P masses of the Main Body during late September and early October. This was not due to the mixing of hypolimnetic waters into the epilimnion. An increase in the total P content of the water column of this degree could only be due to additional inputs or reduced outputs of phosphorus. This period was relatively dry with limited tributary flow and the algal numbers and types did not change drastically during this time. Thus, the primary cause of this increase was most likely the release of phosphorus from the sediments that were formerly within the hypolimnion of the lake.

As discussed by Lee _et al_. (1977) one of the primary factors governing phosphorus exchange from sediments is mixing processes. It would be expected that the amount of phosphorus released when the thermocline was present would be reduced because of the relatively poor transfer of wind-induced currents from the epilimnion to the hypolimnion. With fall overturn the wind-induced currents could more effectively mix the sediments and thereby bring about the release of phosphorus. This is supported by the fact that coincident with the loss of the thermocline in late September, there was an increase in turbidity in the Main Body, i.e, decreased Secchi depths, which could only arise from wind-induced mixing of the sediments under the thermocline. These data strongly support the findings of Lee _et al_. (1977), that aerobic conditions in the water column can result in significant release of phosphorus from the sediments.

In addition to phosphorus release from the sediments, the increased total P mass in the water column most likely reflects a reduced algal uptake of phosphorus and reduced $CaCO_3$ precipitation, both of which would reduce the rate of removal of phosphorus from the water column to the sediments.

Examination of Figure 78, which shows the soluble ortho P mass, gives no clues to the cause of the January algal bloom. Throughout the fall there was a steady decrease in both the total P and the soluble ortho P masses. During this time the algal numbers in the lake were low and generally declining, and there was a relatively small input of phosphorus from the tributaries due to low flow conditions. It is evident that the release of phosphorus from the sediments during late September-early October did not continue throughout the fall at a sufficiently high rate to cause the overall total P and soluble ortho P to increase during this period.

Figures 79 and 80 present the total P and soluble ortho P masses in Rowlett Arm during the study period. Some of the same patterns are seen in Rowlett Arm as in the Main Body. The week-to-week variability in this part of the lake was much greater, probably due to the shallowness of the water body, allowing the wind to readily stir the sediments into the water column.

IMPLICATIONS OF RESULTS FOR WATER QUALITY MANAGEMENT IN NORTH CENTRAL TEXAS IMPOUNDMENTS

The primary justification for support of this study stemmed from the need for information on the approach that should be used for management of water quality in the impoundments in the north central Texas area. NCTCOG, through its Public Law 92-500 Section 208 planning effort, must develop the information necessary to maintain and, where possible, improve water quality in its area of jurisdiction. This study has been limited to Lake

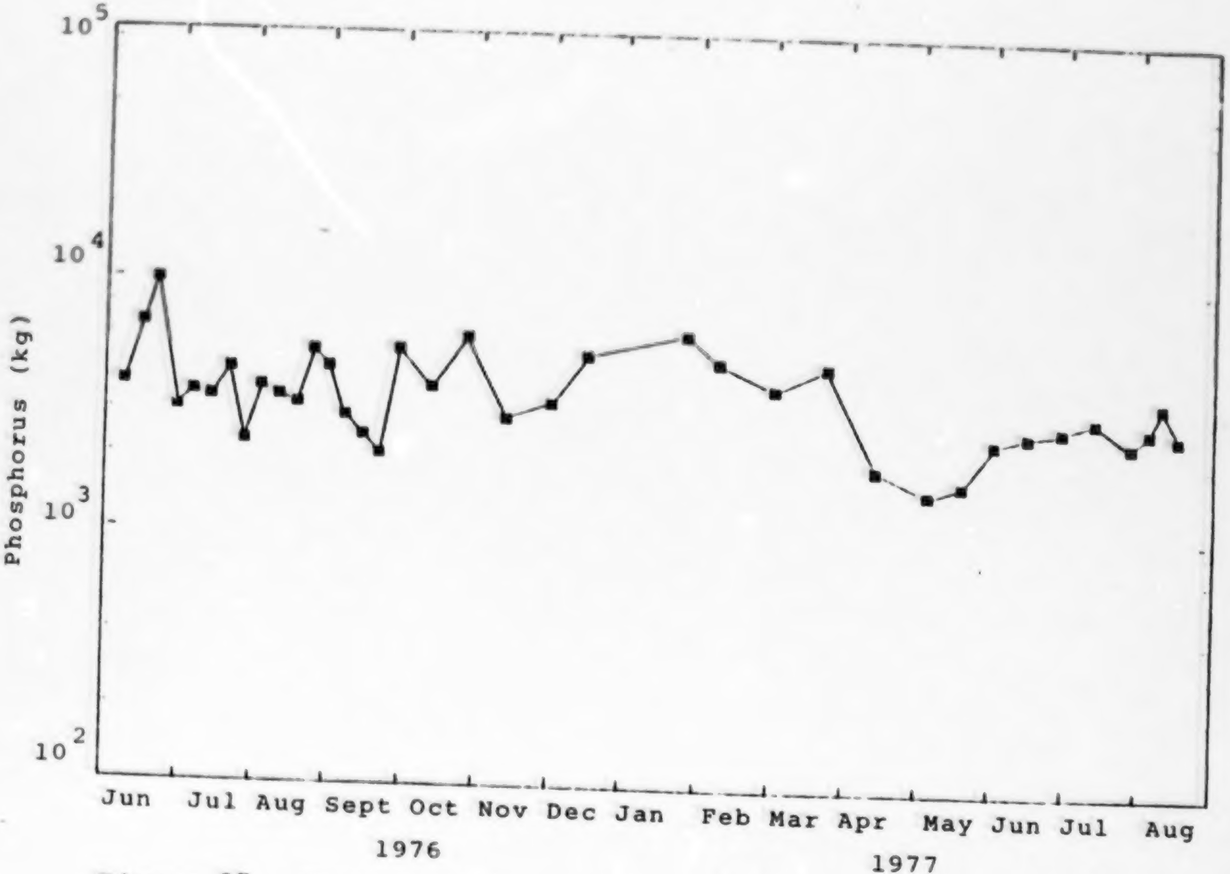

Figure 87. Total phosphorus mass - Rowlett Arm

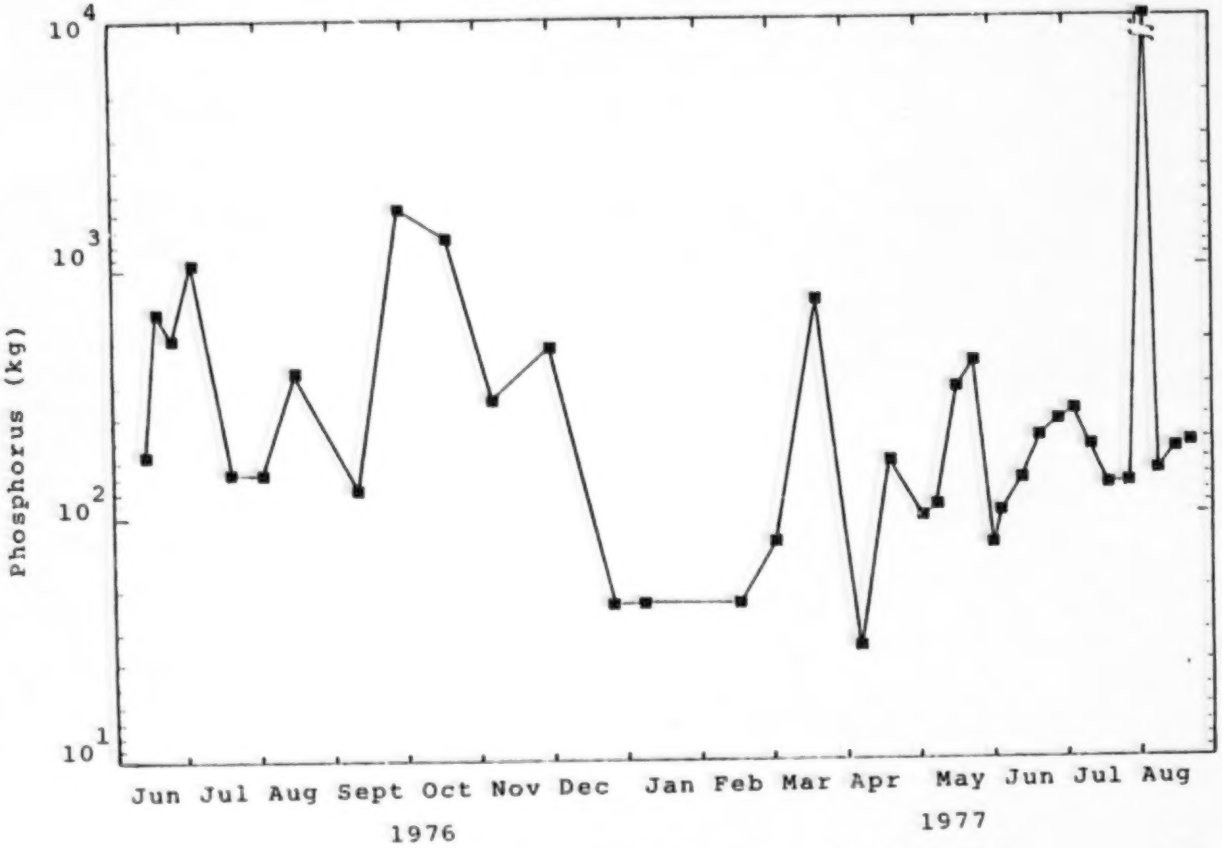

Figure 88. Soluble orthophosphate mass - Rowlett Arm

Ray Hubbard due to funding restrictions, but it is believed that the findings on this lake are applicable to many other water bodies in the region. The proposed management program for Lake Ray Hubbard would, with appropriate modifications for the characteristics of a water body and its watershed, be applicable to water bodies throughout this area.

Almost without exception, all of the impoundments constructed in the NCTCOG area are having problems with excessive growths of algae with concomitant problems of excessive tastes and odors in domestic water supplies. In the Main Body of Lake Ray Hubbard these problems are related to the input of available phosphorus to the lake. It is likely that similar results would be found for the other water bodies in this region. Further, even if studies of the Lake Ray Hubbard type were conducted on other water bodies and showed that nitrogen was the key limiting element controlling algal growth, the approach toward maintaining or possibly enhancing water quality would likely still be that of controlling phosphorus. This situation stems from the fact that, in general, 90 percent phosphorus removal from domestic wastewaters can be readily achieved for less than a cent per person per day for the population served. Nitrogen removal from domestic wastewaters is somewhat more expensive. Further, nitrogen is often derived to a much greater extent from diffuse sources such as urban and rural runoff and the atmosphere than phosphorus is. Both nitrogen and phosphorus are extremely difficult to control from non-point sources. The most important recommendation to NCTCOG arising from this study is to control the algal-available phosphorus input to the water bodies in this region to the maximum extent possible.

It is important to stress the control of algal-available phosphorus versus total phosphorus. Far too often, governmental regulatory agencies focus on total phosphorus control when, a substantial part of the phosphorus, especially from land runoff, is unavailable to support algal growth. This can create a situation where large amounts of funds are spent on phosphorus control with little improvement in water quality. It is important to assess the magnitude of the expected improvement in a water body that can arise from a phosphorus control effort. In some water bodies phosphorus does not control algal growth and the initiation of phosphorus control efforts cannot result in causing phosphorus to limit algal growth. Under these conditions funds are spent for phosphorus control with limited impact on water quality in the water body. It is therefore important that any nutrient control program be critically examined to insure that the desired benefits of water quality will be attained. It is recommended that the OECD eutrophication nutrient load-eutrophication modeling approach, originally developed by Vollenweider and extended by Rast and Lee, be used to predict the improved water quality that would arise from a particular nutrient control program. It is imperative that the regulatory agencies conduct or support the studies necessary to properly define the actual water quality problem that exists in a particular water body, and then obtain the technical information needed to evaluate alternate methods of control. Then, based on this information, the most cost-effective method for controlling the problem to the maximum extent possible with the available financial resources can be developed.

The Lake Ray Hubbard study has provided technical data which support the opinions of fishermen, boaters, and others who use the lake: there are significant differences in water quality between each part of the lake. Rowlett Arm is the most heavily "polluted". The upper part of the East Fork Arm which receives treated domestic wastewater from Rockwall would rank next in terms of degree of pollution. Muddy Creek Arm has water quality characteristics similar to the upper East Fork Arm. The lower part of the East Fork Arm, near Station 3, and the bay at the confluence of Rowlett and Muddy Creek Arms would be classified next in terms of degree of pollution. The Main Body would be classified as the least polluted area of the lake.

Recreation is an important use of Lake Ray Hubbard, but it was constructed by the Dallas Water Utilities for the purpose of providing an additional domestic water supply. In general, it is found that impairment of recreational use usually results in impairment of use for domestic water supply purposes as well. There is certainly not a one-to-one correlation between the two, but in general, the growth of algae to a sufficient extent to impair recreational use almost always significantly impairs the use of the water as a domestic water supply. The City of Dallas is already finding that on occasion the raw water quality in Lake Ray Hubbard is sufficiently degraded to cause significant problems in the use of this water for domestic water supply purposes. For the past three years, DWU has had to stop using Lake Ray Hubbard as a water supply source during the summer because of the severe taste and odor problems associated with this water after treatment.

The impairment of recreational use of a water body as a result of eutrophication does not always lead to tastes and odors in finished drinking water. Three factors must be considered in evaluating the relationship between eutrophication and tastes and odors in treated water. One of these is that there are many different types of algae, only some of which excrete organic compounds which cause tastes and odors. A review on this topic is provided by Palmer (1959). Another factor is that there is considerable support for the idea that the taste and odors of the type experienced in Lake Ray Hubbard are not the direct result of algal excretion of organic materials, but are due to excretion of organics by other organisms such as the actinomycetes. Actinomycetes were not measured throughout the study, but from the measurements made, there appears to be little relationship between the threshold odor number and the actinomycetes counts in the water. This does not necessarily mean that actinomycetes do not cause the odors. There is sufficient evidence that they play some role, so they must be considered as a potential source of odors. The factors controlling the growth of actinomycetes in natural waters are not well understood, but it appears that they are usually associated with more eutrophic waters. However, actinomycetes-like odors have developed in domestic water supplies in which the raw water source would not be considered eutrophic or excessively fertile.

The third factor that must be considered is the treatment method used by the water utility. It is well known in the water utility industry that certain types of treatment methods can greatly reduce and in some instances eliminate taste and odor problems. The domestic water supply industry in the US tends to be rather traditional and conservative in its approach to initiating new treatment methods which increase the cost of treatment. It is sometimes said that it is not possible to control the excessive tastes and odors of the type that the City of Dallas is experiencing in their attempts to use Lake Ray Hubbard water as a raw water supply during late summer. DWU experiences indicate that this is not the case. The City of Dallas was fortunate in having Lake Tawakoni available as an alternate raw water supply source for the East Side Water Treatment Plant. According to discussions with F. Stone of the Dallas Water Utilities, Lake Tawakoni also has significant tastes and odors in the treated water derived from it if conventional treatment methods are used. However, Mr. Stone has found that by greatly increasing the chlorine dose in the transmission pipe between Lake Tawakoni and the East Side filtration plant, they have been able to essentially eliminate (burn up) these tastes and odors. This approach is not conventional treatment but is a practice that could be instigated at the East Side filtration plant for Lake Ray Hubbard water, if a chlorine contact tank were installed.

Another example of the potential importance of the treatment method in determining the impact of excessive fertilization of Lake Ray Hubbard on taste and odors in the treated waters derived from this lake, in the use of activated carbon for removal of tastes and odors. There are two methods by which activated carbon is used. The least expensive and simplest method involves the use of powdered carbon which is added during normal treatment processes. There are times when water plant operators experience situations where they cannot add powdered carbon rapidly enough to completely eliminate a severe "outbreak" of tastes and odors. The other approach involves the use of activated carbon beds. It has been known for many years that there are few taste and odor problems in water supplies which cannot be eliminated by the proper use of activated carbon beds. This approach would likely eliminate the taste and odor problems that the DWU East Side Water Treatment Plant has experienced; however, it would also significantly increase the cost of treatment.

Thus far, the DWU managers and water utility managers across the US have chosen not to initiate the more expensive approach of activated carbon beds to eliminate tastes and odors. In a sense, a judgment has been made that taste and odor problems are important but are usually too short-lived to justify activated carbon treatment. Some members of the public complain about the bad tasting water during these episodes, but many accept and are willing to put up with it to save a few cents per thousand gallons of water used. The role of the water treatment plant practice itself in stimulating tastes and odors is not clearly understood at this time.

The data gathered in this study show that the tastes and odors of the finished drinking water are poorly correlated with the total numbers or types of algae. There are trends in the data which seem to show an association between the threshold odor number and number of blue-green algae. However, a correlation of these two would have a poor correlation coefficient. This is expected because the blue-green algae are made up of many different species, each with its own odor characteristic. Even if all the odors in Lake Ray Hubbard were due to blue-green algae, one would still expect a relatively poor correlation between the number of blue-green algae and the threshold odor number of the water. In addition to being of varying types, the age of the blue-green algal population and a variety of other factors would be influential in determining this relationship.

It is obvious that a two-pronged approach should be used in developing the water quality management program for Lake Ray Hubbard. One of these involves the control of phosphorus to the maximum possible extent. The second involves the control of taste and odor problems associated with odorous compounds in the raw water which are not removed by conventional water treatment processes. If the taste and odors are judged to be of sufficient problem to cause remedial action to be taken after control of the readily controllable phosphorus inputs has been achieved, then consideration should be given to initiating modified water treatment practices to eliminate the residual tastes and odors.

One of the factors which may play an important role in future water treatment practices in the US is the formation of trihalomethanes. These "chloroform" type compounds are of great concern to the US Environmental Protection Agency and many other individuals associated with domestic water supply within the US. The trihalomethanes which are known to cause cancer in some organisms, are present in essentially all finished waters derived from surface water supplies. They are formed from reactions between what are termed "precursor compounds", which are organic materials derived from the decay of terrestrial and aquatic plants. Municipal and some industrial wastes probably contribute significantly to the production of organic materials which lead to the formation of the precursors. There is increasing evidence that the eutrophication of natural waters leads to increased precursor formation with a concomitant increased concentration of trihalomethanes in finished waters derived from eutrophic water supplies.

It is difficult to assess the significance of trihalomethanes in domestic water supplies. This is an area of considerable discussion and debate in the water supply industry today. The concentrations found in domestic water supplies, in general, are very low compared to other sources of trihalomethanes. The significance of small amounts of known carcinogens in water, food, and other materials is under active debate at this time. In the opinion of the authors, it is fair to state that there is some small, but definite, risk associated with consumption of drinking water which contains trihalomethanes but the magnitude of this risk is unknown. It appears, however, that its significance is small compared to other sources of the same or similar compounds, such as food or air. As a result, the domestic water supply industry in the US is in a quandary as to what approach to take with respect to removal of trihalomethanes. It is known that these materials can be readily removed through the use of activated carbon columns. There are many concomitant benefits associated with this treatment since activated carbon beds also remove many other potentially significant compounds including the odorous compounds of concern to DWU. There are other compounds in treated waters that may be hazardous to human health which are likely to be removed on activated carbon beds.

The US Environmental Protection Agency is currently developing regulations covering trihalomethanes in treated domestic water supplies. The magnitude of the numbers are yet to be defined, but it is likely that there will be requirements for removal of trihalomethanes from those water supplies that currently have the highest concentrations. The critical values of trihalomethanes requiring treatment will be lowered over the years until essentially all of the US population is consuming water low in trihalomethanes. It may be that in time, sufficiently strict regulations will be adopted by the US EPA so that DWU, as well as other water utilities in the region, will have to initiate treatment practices which would remove the trihalomethanes. Should this occur, the taste and odor problems caused by excessive fertilization of the raw water supplies would be considerably diminished.

One of the important issues in developing a control strategy for excessive fertilization of Lake Ray Hubbard, as well as other water bodies in the north central Texas region, is the diversion of domestic wastewaters around the water bodies to the streams below. It would be relatively simple to construct discharge pipes to take the wastewaters from the Garland-Rowlett and at somewhat greater expense, the Plano sewage treatment plants, to the East Fork of the Trinity River, below Lake Ray Hubbard. This approach has been used in other parts of the country as a means of controlling excessive fertilization. There is considerable reluctance to take this approach in the Dallas area since it means that a considerable amount of potentially available water would be diverted around Lake Ray Hubbard. During periods of drought and especially under conditions of increased growth, the City of Dallas and the suburbs could encounter a situation similar to that of the 1950's in which there was insufficient water available to meet the public's needs. There are many who feel that the wastewaters should be treated to the degree necessary to minimize their adverse impact on water quality within Lake Ray Hubbard, and thereby retain the water in a form available for use for domestic water supply purposes. This represents a form of wastewater recycle, utilizing the lake for further purification and dilution of the wastes prior to its use as a domestic water supply for the DWU. This is typically practiced throughout the US, although in most cases the practice involves one city's utilizing an upstream city's wastewaters as a source of water supply. In the case of

Lake Ray Hubbard, it is, to some extent, a recycle of wastewaters of the city from which the water was originally derived.

Phosphorus Removal from Domestic Wastewaters

The control of excessive fertilization of lakes and impoundments could be directed toward limiting nitrogen and phosphorus from a wide variety of sources; however, as discussed by Lee (1973b), it is usually restricted to controlling phosphorus in domestic wastewaters. This is normally done by removing phosphate at the sewage treatment plants by advanced wastewater treatment or by completely or partially banning the use of phosphate detergents in the watershed. The basis for a phosphate detergent ban is the claim that the phosphorus used as a builder in detergent formulations contributes significantly to the total phosphorus content of domestic wastewaters. Around 1970, the total P content of domestic wastewaters was approximately double what it would have been if phosphorus had not been used as a builder in synthetic detergents. The phosphorus content of domestic wastewaters has been significantly decreased since the early 1970's as a result of a decrease in the phosphorus content of detergents. Today, approximately 35 percent of the total phosphorus present in US domestic wastewaters is derived from the household use of detergent formulations containing phosphorus. A complete detergent phosphate ban in the Lake Ray Hubbard watershed would be expected to reduce the phosphorus loading from domestic wastewater sources by 35 percent which would result in a 15 percent reduction in the total loading to the lake. As discussed in another section of this report, the impact of this reduction on water quality in the Main Body of Lake Ray Hubbard would not be measurable.

It is generally agreed by those who have carefully reviewed the matter that if limiting phosphorus in domestic wastewaters will have a beneficial effect on water quality in a water body, then the proper approach to take is that of initiating phosphorus removal at the domestic wastewater treatment plant. Phosphorus control in domestic wastewater effluents has received considerable attention in the Great Lakes region of the US and Canada. Several years ago, the US and Canada adopted the policy of requiring advanced waste treatment processes for domestic wastewater treatment plants that were discharging to the Great Lakes or their tributaries. This reduced the phosphorus content of the wastewater treatment plant effluents to about 1 mg P/l, i.e., about 90 percent reduction from the concentrations in the influent waters to the treatment plants. There are many domestic wastewater treatment plants located in Canada, the US, and Sweden with advanced wastewater treatment processes which are routinely achieving 80 to 90 percent removal of phosphorus. The cost of this practice is on the order of less than one cent per person per day for the population served, for treatment plants with flows greater than about one million gallons per day.

It is concluded, based on the experience of the authors, that phosphorus removal in domestic wastewaters for the cities of Garland-Rowlett, Plano, and Rockwall is technically and economically feasible. Adoption of such practices with proper treatment plant operation can readily result in an 85 percent reduction in the phosphorus added to Lake Ray Hubbard from domestic wastewater sources. This would result in a 39 percent reduction in the total annual phosphorus load to the lake. The impact of this reduction on water quality within the lake is discussed in another section of this report.

One of the areas which sometimes receives attention with respect to detergent phosphate bans is that of the reduced costs associated with the removal of phosphorus at the domestic wastewater treatment plant as a result of reduced total phosphorus load to the plant. This topic has been reviewed in detail in the Great Lakes region, both in the US and Canada. It has been found that if iron or aluminum salts are used for phosphorus removal in the domestic wastewater treatment plants, the cost increases due to greater chemical use with increasing phosphorus content of the domestic wastewaters. However, with the use of lime as a phosphorus removal chemical, the cost of removal is independent of the phosphorus concentration. At this time, the most economical approach for phosphorus removal appears to be that of the use of iron salts as precipitating chemicals. In a statement before the International Joint Commission for the Great Lakes in July 1977, the US EPA representatives indicated that while there is an increased cost associated with higher phosphorus levels in domestic wastewaters when iron or aluminum salts are used as a precipitating chemical for phosphorus, the cost to the consumer of using non-phosphate-built detergents of the type that are in use today in the US is far greater than the costs of removing this additional phosphorus at the treatment plant. The increased consumer costs are due to shortened washing machine life and reduced useful life of clothes.

In summary, the best approach to follow today for control of phosphorus in domestic wastewaters is that of removal of phosphorus in the domestic wastewater treatment plant. This is a much more cost-effective method of achieving phosphorus control than detergent phosphate bans. In addition, a number of other concomitant benefits are derived from removal of phosphorus in the domestic wastewater treatment plant. For many domestic wastewater treatment plants, phosphorus removal also results in increased removal of BOD and suspended solids. Another benefit which will be of importance in the future is that advanced treatment processes designed to remove phosphorus also remove many other potentially hazardous chemicals from domestic wastewaters.

Control of Phosphorus from Diffuse Sources

It is difficult, if not virtually impossible to control phosphorus from diffuse sources such as land runoff and the atmosphere. Septic tank wastewater effluents are often considered as a source of phosphorus in control programs. As discussed in a previous section of this report, it appears that the use of septic tank systems as a method of wastewater disposal within the Lake Ray Hubbard watershed is not contributing significantly to the phosphorus load to Lake Ray Hubbard which could stimulate algal growth in the lake.

This study has shown that there are significant amounts of nitrogen and phosphorus contributed to Lake Ray Hubbard from land runoff during periods following storms. Therefore, it is important to consider the potential significance of these land runoff-derived nutrients. A considerable amount of the phosphorus added to Lake Ray Hubbard from land runoff is in a particulate form. Only a small fraction of this particulate phosphorus would likely become available to stimulate algal growth in the arms of Lake Ray Hubbard and only a fraction of what is added to the arm would become available to stimulate algal growth in the Main Body of the lake. While a reliable estimate of this fraction is not available at this time, it is certainly no greater than a few percent of the total particulate phosphorus added to the lake. Using the approach developed by Cowen and Lee (1976a), the best estimate available today, using 20 percent of the particulate phosphorus as being potentially available for support of algal growth in Lake Ray Hubbard, indicates that the total available phosphorus added to the Rowlett Arm of the lake would be the soluble orthophosphate plus 20 percent of the particulate phosphate. It has been found that about 7 percent of the phosphorus added to the Rowlett Arm of the lake reaches the Main Body. Part of the P removal is related to particulate phosphorus sedimentation. Therefore, control programs directed toward reducing phosphorus from land runoff are likely to have very little impact on water quality within the Main Body of Lake Ray Hubbard because of the relatively small change in the available phosphorus load that would occur in the Main Body of the lake.

It is expected that most of the phosphorus inputs from atmospheric sources would be added in a particulate form, and little of this particulate P would become available to support algal growth within the lake. Based on studies in other parts of the country, a substantial part of the particulate forms of nitrogen will likely become available to support algal growth in the lake. However, since overall the lake is phosphorus limited, the additional availability of nitrogen from particulate forms appears to be of minor consequence in influencing water quality. At certain times of the year Rowlett Arm is nitrogen-limited but it is likely that reduced light penetration due to the turbidity in the water column plays an important role in controlling algal growth in this part of the lake. Therefore, the particulate nitrogen forms may not be important in stimulating algal growth in this part of the lake.

The current IJC Research Advisory Board activities concerned with eutrophication control in the Great Lakes should be of interest to NCTCOG in that, for the first time, a substantial attempt is being made to develop cost-effective phosphorus control programs for water bodies in a large geographical area of the US. There have been similar attempts in other parts of the US in the past but they have been conducted on a much more restricted basis. The IJC study is considering the control of phosphorus from a wide variety of sources, including domestic wastewaters, land runoff, and the atmosphere. A task force consisting of experts involved in various aspects of nutrient control is being appointed to work with the Research Advisory Board and the Expert Committee on Engineering and Technology to develop a phosphate management strategy for the Great Lakes. This task force is supposed to begin operation early in 1978 and will be considering the technology, economics, and social impact of various alternatives for phosphorus control.

It will be able to draw heavily from the past several years' multi-million dollar research effort that has been conducted in several of the Great Lakes states and in Canada, devoted to the impact of land use on Great Lakes water quality. This study, commonly called PLUARG, represents one of the most comprehensive studies of this type ever conducted. Some of the more important aspects of the PLUARG activities include a definition of the amounts of aquatic plant nutrients derived from various types of land use, studies on the amounts of nutrients available to support algal growth, and the control of nutrients from land runoff sources which is an extremely important, essentially unstudied area. The latter includes working with the farmers and others in attempting to obtain adoption of altered land-use practices to minimize nutrient, and especially phosphorus transport. Those responsible for water quality management in the NCTCOG area should carefully review the results of the PLUARG study and the findings of the IJC Research Advisory Board task force devoted to phosphate management strategy. This work will likely serve as a basis for similar types of programs in many other parts of the US.

Management Control Strategy

In their 208 planning effort, NCTCOG must decide if the municipalities in this region should be required to initiate advanced waste treatment practices which will bring about phosphorus removal in domestic wastewaters discharged to Lake Ray Hubbard. The decision as to whether to require this type of practice at this time, or at some time in the future, should be based on expected benefits due to improved water quality in Lake Ray Hubbard arising from phosphorus control in the domestic wastewater effluents. In making this decision, the fact that both Plano and Garland-Rowlett have initiated or soon will initiate, modified wastewater treatment practices which can be readily modified to effect substantial phosphate reduction in the effluents, should be considered. NCTCOG should work with both of these municipalities and their consulting engineers to determine the actual increase in costs that would occur to the municipalities as a result of operating their new treatment plants for phosphate removal. It is possible, because the cities already are, or soon will be, operating advanced waste treatment plants, that the additional cost would be relatively small compared to initiating this type of treatment at a conventional domestic wastewater treatment plant. It should be noted that many consulting engineering firms in various parts of the US are not familiar with normal phosphate removal practices in domestic wastewater treatment plants. If the consulting engineering firms used by these two cities have limited experience in this area, then consulting engineers who have been active in the Great Lakes region should be contacted for their advice on this matter.

There is frequently confusion in the minds of the public and public officials concerning effluent standards placed on wastewater discharges as they relate to eutrophication problems. The conventional wastewater discharge limitations of a certain number of milligrams per liter of suspended solids and BOD have little or nothing to do with the impact of the wastewaters on lakes and impoundments with respect to eutrophication. In the Lake Ray Hubbard Basin, the Texas Water Quality Board, now the Texas Department of Water Resources, has specified BOD and suspended solids limits on various treatment plants discharging to Lake Ray Hubbard tributaries. In some instances, these limits are fairly strict compared to normal BOD and suspended solids reductions achieved in conventional wastewater treatment plants. These strict limits will have limited impact on the eutrophication problems of Lake Ray Hubbard since suspended solids and BOD discharged by the wastewater treatment plants do not control the rate of eutrophication. The benefits from the low suspended solids and BOD in the wastewaters from the treatment plants will be seen in the streams that receive the wastewaters and not to any significant degree in the lakes. In order to protect the lakes, it will be necessary to initiate nutrient control at the treatment plants. This should be done if it is found that the potential benefits of these control efforts will outweigh the costs associated with the efforts.

APPROACH TO WATER QUALITY MANAGEMENT IN THE NCTCOG AREA

Definition of Problem

Those responsible for water quality management should clearly define what problems or potential problems exist or will exist in the water body under consideration. It is important that the management be directed toward a specific problem or group of problems. If controlling nutrient input is the approach being taken, it should be understood that there may be a limited relationship between nutrient input and a specific water quality problem such as tastes and odors in domestic water supplies. In some instances, it may be necessary to do additional research before the most cost-effective program can be developed for management of excessive fertilization-related problems in impoundments in this region. This research would have to focus on the relationships between chlorophyll, algal numbers and types, and taste and odor problems in treated water. It would have to include a detailed investigation of the impact of water treatment tastes and odors in the treated water.

Another problem that could be present in the NCTCOG area which does not appear to be present now is that of floating macrophytes. If this problem develops in this region, then a different management strategy may be required since there is little known about nutrient load-response relationships for floating or attached macrophyte problems. If the water quality problems of concern or potential concern are related to planktonic algal growth to any significant extent, then the approach presented below should be used.

Determination of Nutrient Sources

The watershed of the particular water body should be defined from topographic maps. If areal photography of the watershed has not been made within the last five years, the watershed should be flown again and an areal photograph with sufficient resolution to define the dominant types of land use should be obtained. The dominant types of land use should be restricted to cropland, pasture, forest, and urban area. There is no need for further definition at this time since the amounts of nutrients derived from further subdivisions of each of these major categories are not known. It is not recommended that efforts be made to better define nutrient loads from each of the sub-categories since the variability of nutrient loads as a result of climatic conditions will probably be greater than the differences between various land use types within a specific category of this type.

Of particular importance in defining nutrient sources is the discharge of wastewaters to tributaries of the water body. If periodic measurements of the flow and nutrient content discharged from a sewage treatment plant are not available, then the approach used in this study for estimating the nutrient load from the Rockwall and Plano sewage treatment plants should be adopted. It is appropriate to assume that in this region the flow of domestic wastewaters would be on the order of 380 liters per capita per day and that the phosphorus load would be approximately 1.1 kg/person/yr after conventional treatment, and the nitrogen load from a sewage treatment plant will be on the order of 2.7 kg/person/yr. The estimated nutrient loads as a result of direct precipitation and fallout on lakes should be made in accord with the procedures used in this report. Until better estimates are available, it is recommended that the following values be used.

Atmospheric Loading Coefficients
($g/m^2/yr$)

	Total Phosphorus	Total Nitrogen
Rainfall	0.02	0.8
Dustfall	0.08	1.6
Total Atmosphere	0.1	2.4

It is recommended that once land use is known, nutrient export coefficients of 50 kg total $P/km^2/yr$ and 350 kg total $N/km^2/yr$ be used to estimate the nutrient loading from land runoff.

It should be determined if there are any potentially significant nutrient sources such as animal feedlots in the watershed. If a potentially high nutrient source is found, then it will be necessary to do a study to ascertain the amounts of nutrients derived from that particular source.

If the watershed contains another water body upstream from the water body of interest, which has a hydraulic residence time greater than a few weeks, then the nutrient load to the upper water body should be reduced by about 80 percent in order to determine the nutrient load to the lower water body.

The USGS should be able to provide assistance in estimating the amounts of water derived from various types of land use in the region. Usually, the engineer's report associated with the design of the impoundment would provide sufficient information on the hydrologic characteristics of the watershed to meet the needs for developing management control programs.

Characteristics of the Impoundment

The important characteristics of the impoundment that must be available are its mean depth and hydraulic residence time. Information of this type should be readily available from the engineer's report that was used in preparation of the proposal for the construction of the impoundment. If a major part of the nutrients enter into parts of the impoundment which have somewhat restricted circulation with the Main Body (e.g., the arms of Lake Ray Hubbard), the total nutrient load to the Main Body should be reduced by at least 50 percent. Although each impoundment will behave differently, if the flows during the summer months are low enough to produce hydraulic residence times in the arms of approximately a month or more, then it is likely that substantial nutrient removal will take place within those parts of the impoundment. Once the characteristics of the impoundment have been defined, it is possible to make estimates of the potential for nitrogen or phosphorus limitation in the impoundment based on the nutrient loads.

Insufficient information is available at this time to be certain that phosphorus will be the key limiting element in the main parts of all water bodies in this region. It is likely that some of the arms will behave similarly to Rowlett Arm, where at times there are sufficiently large amounts of phosphorus entering the arm to cause nitrogen limitation. This would most likely be encountered if there is a large input of wastewaters to the arm.

In order to determine whether phosphorus is limiting or can be made limiting, it is necessary to have some measurements of available forms of nitrogen and phosphorus during the critical growing season, i.e., late summer. Measurements made at approximately monthly intervals during June, July, August, and September should be adequate to define whether nitrogen or phosphorus is limiting. Measurements of soluble orthophosphate, total phosphate, ammonia, nitrate, nitrite, organic nitrogen, and chlorophyll in whole water samples should be made. Water samples should be taken about 1 m below the surface for these measurements. It is important that the collection and analyses of the samples are done in a manner which preserves the initial distribution of the forms of nitrogen and phosphorus in the samples. Far too often studies of this type produce essentially meaningless results because the samples are analyzed months after collection.

The Secchi depth of the water during the summer and fall should be measured at about two-week intervals and certainly no less than monthly. This measurement provides extremely important information for judging changes in water quality.

If there are water quality problems at other times of the year which are related or potentially related to algal growth, measurements should be made during these times as well. From the above information, someone knowledgeable in the environmental chemistry of nutrients and the nutrient load-response relationships in water bodies can make a reasonably good assessment of the factor limiting algal growth in the water body and the important sources of this limiting agent.

Projected Water Quality

Using the OECD eutrophication modeling approach presented in this report and the above-listed information, it would be possible to predict with a reasonable degree of reliability, the impact of various nutrient control programs on water quality in any water body of interest. The potential benefits from removing 85 percent of the phosphorus in domestic wastwaters should be the primary focal point of the evaluation of alternate management strategies. This can be done through the use of the OECD eutrophication models following the procedures presented in this report.

The next phase of the evaluation should include an estimate of population increase in various parts of the watershed over 10 to 20 years and then, using typical per capita concentrations of phosphorus and nitrogen, such as the ones used in this report, estimate the increased phosphorus and nitrogen loads from the increased population. This load in turn can be used to estimate the changes in water quality that will occur as a result of increased population and manipulation of the phosphorus load associated with this population.

It should be stressed that this approach is directed toward controlling the overall "greenness" of the water body, where "greenness" relates to the chlorophyll content of the algae. In general, chlorophyll is a reasonably good indicator of general water quality characteristics as they relate to recreational and water supply use. There are, as discussed in the text, some problems with this approach. As long as there is a relatively poor understanding of the cause of the major taste and odor problems in the impoundments such as Lake Ray Hubbard, one cannot automatically assume that reducing the chlorophyll content will reduce the taste and odor problems to a similar degree. Based on the current state of knowledge, however, it is felt that this is the best approach to take at this time and that additional research should be initiated to better understand the relationship between the nutrient load to a water body and the impairment of use for domestic water supply purposes.

SECTION X

REFERENCES

American Public Health Association, American Water Works Association, Water Pollution Control Federation. 1965. Standard Methods for the Examination of Water and Wastewater. 12th ed. New York, American Public Health Association.

American Public Health Association, American Water Works Association, Water Pollution Control Federation. 1971. Standard Methods for the Examination of Water and Wastewater. 13th ed. New York. American Public Health Association. 874 p.

American Public Health Association, American Water Works Association, Water Pollution Control Federation. 1976. Standard Methods for the Examination of Water and Wastewater. 14th ed. New York, American Public Health Association. 1193 p.

Brezonik, P.L. and Lee, G.F. 1966. Sources of Elemental Nitrogen Sink in Fermentation Gases. Air & Water Poll. 10: 145-160.

Brezonik, P.L. and Lee, G.F. 1968. Denitrification as a Nitrogen Sink in Lake Mendota. Environ. Sci. and Tech. 2:120-125.

Clesceri, N.L. and Lee, G.F. 1965a. Hydrolysis of Condensed Phosphates. I: Non-Sterile Environment. Air & Water Poll. 9:723-742.

Clesceri, N.L. and Lee, G.F. 1965b. Hydrolysis of Condensed Phosphates. II: Sterile Environment. Air & Water Poll. 9:743-751.

Cowen, W.F. and Lee, G.F. 1976a. Phosphorus Availability in Particulate Materials Transported by Urban Runoff. Water Poll. Cont. Fed. 48:580-591.

Cowen, W.F. and Lee, G.F. 1976b. Algal Nutrient Availability and Limitation in Lake Ontario During IFYGL. Part I. Available Phosphorus in Urban Runoff and Lake Ontario Tributary Waters. US EPA 600/3-76-094a, Environmental Research Laboratory, Duluth. 218 p.

Cowen, W.F., Sirisinha, K. and Lee, G.F. 1976. Nitrogen Availability in Urban Runoff. Water Poll. Cont. Fed. 48:339-345.

Dallas Morning News. 1975. Texas Almanac and State Industrial Guide. A. H. Belo Corporation. 704 p.

DiToro, D.M., O'Connor, D.J. and Thomann, R.V. 1971. A Dynamic Model of the Phytoplankton Population in the Sacramento-San Joaquin Delta. Advances in Chemistry. 106:131-180.

Edmondson, W.T. 1972. Nutrients and Phytoplankton in Lake Washington. In: G.E. Likens (ed.) Nutrients and Eutrophication: The Limiting Controversy, Limnol. Oceanogr., Special Sym., Vol. I, pp. 172-193.

Eppley, R.W. 1972. Temperature and Phytoplankton Growth in the Sea. Fishery Bulletin. 70 (4):1063-85.

Forrest and Cotton, Inc. Consulting Engineers. 1960. Design Memorandum No. 1, Design Floods and Freeboard Requirements, Forney Dam and Reservoir. April 18, 1960. 27 p.

Gardner, W.S. and Lee, G.F. 1975. The Role of Amino Acids in the Nitrogen Cycle of Lake Mendota. Limnol. Oceanogr. 20: 379-388.

International Business Machines. 1972. IBM System/360 Continuous System Modeling Program User's Manual. IBM Publication GH20-0367-4. 76 p.

Jones, R.A. and Lee, G.F. 1977. Septic Tank Disposal Systems as Phosphorus Sources for Surface Waters. US EPA Report. Robert S. Kerr Environmental Research Laboratory. (In press).

Jones, R.A. and Lee, G.F. Detergent Content of U.S. Domestic Wastewaters. (Submitted for publication).

Lee, G.F. 1969. Analytical Chemistry for Plant Nutrients. In: Proceedings of Symposium on Eutrophication, National Academy of Sci., Madison, Wisconsin, pp. 646-658.

Lee, G.F. 1970. Factors Affecting the Transfer of Materials Between Water and Sediments. Literature Review No. 1, Eutrophication Information Program, University of Wisconsin, Madison. 50 p.

Lee, G.F. 1973a. Eutrophication. Transactions of the Northeast Fish and Wildlife Conference. pp. 39-60.

Lee, G.F. 1973b. Role of Phosphorus in Eutrophication and Diffuse Source Control. Water Research. 7:111-128.

Lee, G.F. 1976. Review of the Potential Water Quality Benefits from a Phosphate Built Detergent Ban in the State of Michigan. Presented at State of Michigan Hearings on Proposed Phosphate Detergent Ban, State of Michigan, December 1976.

Lee, G.F. 1977. Critical Levels of Phosphorus and Nitrogen in Texas Impoundments. Texas J. of Sci. XXVIII:347-350.

Lee, G.F. The Effects of Madison Metropolitan Waste Water Effluent on Water Quality in Badfish Creek, Yahara, and Rock Rivers. Transactions Wisc. Academy Sciences, Arts and Letters. (In press).

Lee, G.F. and Delfino, J.J. 1969. Discussion, Equilibrium and Kinetic Aspects of Inorganic Chemical Quality in Reservoirs. In: Proceedings of Specialty Conference on Current Research into the Effects of Reservoirs on Water Quality. Jan. 1968. ASCE Technical Report No. 17. Dept. Environ. and Water Resources Engr. Vanderbilt Univ. pp. 51-72.

Lee, G.F. and Harlin, C.C. 1965. Effect of Intake Location on Water Quality. Ind. Water Eng. 2:36-40.

Lee, G.F. and Hoadley, A.F. 1967. Biological Activity in Relation to the Chemical Equilibrium Composition of Natural Water. In: Equilibrium Concepts in Natural Water Systems, Advances in Chemistry Series No. 67, Amer. Chem. Soc. pp. 319-338.

Lee, G.F. and Martin, P.H. 1975. Dissolved Gas Supersaturation and Dilution in Thermal Plumes from Steam Electric Generating Stations. Water Research. 9:643-648.

Lee, G.F., Sonzogni, W.C. and Spear, R.D. Significance of Oxic Versus Anoxic Conditions for Lake Mendota Sediment Phosphorus Release. In: Proceedings of International Symposium on Interactions Between Sediments and Fresh Water. 1976. W. Junk, Purdoc, The Hague, pp. 294-306.

Lee, G.F. and Veith, G.D. 1971. Effects of Thermal Discharges on the Chemical Parameters of Water Quality of Eutrophication. In: Proceedings of the International Symposium on Identification and Measurement of Environmental Pollutants, National Res. Council of Canada, Ottawa, Canada, June, 1971. pp. 287-294.

Natrella, M.G. 1963. Experimental Statistics. National Bureau of Standards Handbook 91, Superintendent of Documents.

North Central Texas Council of Governments. 1971. 1970 Fact Book. North Central Texas Council of Governments. Arlington, Texas. 290 p.

North Central Texas Council of Governments. 1977a. Current Population Estimates, 1976. North Central Texas Council of Governments. Arlington, Texas. 6 p.

North Central Texas Council of Governments. 1977b. Analysis of Sewage Load and Flow Factors. North Central Texas Council of Governments. Arlington, Texas. 25 p.

Otsuki, A. and Wetzel, R.G. 1972. Coprecipitation of Phosphate with Carbonates in a Marl Lake. Limnol. Oceanogr. 17:763-767.

Palmer, C.M. 1959. Algae in Water Supplies. Publication No. 657. U.S. Department of Health, Education and Welfare Public Health Service, Cincinnati, Ohio. 88 p.

Rast, W. and Lee, G.F. 1977. Report to US EPA Corvallis Research Laboratory, Corvallis, Oregon on US OECD Eutrophication Study Results. 442 p. (In press).

Richardson, W.L. and Bierman, V.J. 1976. A Mathematical Model of Pollutant Cause and Effect in Saginaw Bay, Lake Huron. US EPA Report EPA-600/3-76-079. Water Quality Research of the Environmental Protection Agency. pp. 38-58.

Rawson, D.S. 1955. Morphometry as a Dominant Factor in the Productivity of Large Lakes. Verh. Internat. Verein. Limnol. 12:164-175.

Shannon, J.E. and Lee, G.F. 1966. Hydrolysis of Condensed Phosphates in Natural Waters. Air & Water Poll. 10:735-756.

Sonzogni, W.C. and Lee, G.F. 1972. Nutrient Sources for Lake Mendota - 1972. Trans. Wisc. Academy Sciences, Arts and Letters. LXII:133-164.

Sonzogni, W.C., Uttormark, P.C. and Lee, G.F. 1976. A Phosphorus Residence Time Model: Theory and Application. Water Research. 10:429-435.

Stauffer, R.E. and Lee, G.F. 1973. The Role of Thermocline Migration in Regulating Algal Blooms. In: Middlebrooks, E.J., D.H. Falkenborg and T.E. Maloney (eds.), Modeling the Eutrophication Process, Workshop Proceedings, Utah State University, Logan, September 6-7, 1973. pp. 73-82.

Stumm, W. and Morgan, J.J. 1970. Aquatic Chemistry. Wiley Interscience. New York. 538 p.

Thomann, R.V., DiToro, D.M., Winfield, R.P. and O'Connor, D.J. 1975. Mathematical Modeling of Phytoplankton in Lake Ontario 1. Model Development and Verification. US EPA Report EPA-660/3-75-005. 177 p.

Thomann, R.V., Winfield, R.P., DiToro, D.M. and O'Connor, D.J. 1976. Mathematical Modeling of Phytoplankton in Lake Ontario. 2. Simulations Using Lake 1 Model. US EPA Report EPA-600/3-76-065. 87 p.

Torrey, M.S. 1972. Biological Nitrogen Fixation in Lake Mendota. Ph.D. Dissertation. University of Wisconsin, Madison, Wisconsin. 437 p.

Torrey, M.S. and Lee, G.F. 1976. Nitrogen Fixation in Lake Mendota, Madison, Wisconsin. Limnol. Oceanogr. 21:365-378.

US Department of Agriculture, Soil Conservation Service. 1969. Soil Survey, Collin County, Texas. Temple, Texas.

US Department of Agriculture, Soil Conservation Service in Cooperation with Texas Agricultural Experiment Station. 1975. General Soil Map, Dallas County, Texas. Temple, Texas. Revised January 1975.

US Department of Commerce, National Oceanic and Atmospheric Administration, Environmental Data Service. 1973. Local Climatological Data. National Weather Service Met. Obsy. Ft. Worth, Texas.

US Environmental Protection Agency. 1974. Methods for Chemical Analysis of Water and Wastes. US EPA Report EPA-625/6-74-003, 298 p.

US Environmental Protection Agency. 1976. Quality Criteria for Water. US EPA-440/9-76-033, 501 p.

US Environmental Protection Agency. 1977a. National Eutrophication Survey Working Paper No. 649. Report on Lake Lavon, Collin County, Texas. EPA Region VI. 16 p.

US Environmental Protection Agency. 1977b. National Eutrophication Survey Working Paper No. 650. Report on Lake Lewisville, Denton County, Texas. EPA Region VI. 18 p.

Vollenweider, R.A. 1975. Input-Output Models, with Special Reference to the Phosphorus Loading Concept in Limnology. Schweiz Z. Hydrol. 37:53-84.

Vollenweider, R.A. 1976. Advances in Defining Critical Loading Levels for Phosphorus in Lake Eutrophication. Mem. Ist. Ital. Idrobio. 33:53-83.

Ward, H.B. and Whipple, G.C. 1959. *Freshwater Biology*. W.T. Edmondson (ed.). John Wiley & Sons, Inc. New York. 1248 p.

QUALIFICATIONS OF THE AUTHORS

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Dr. G. Fred Lee is Professor of Environmental Sciences at the University of Texas at Dallas. Professor Lee is principal investigator and principal author of this report. His Ph.D. dissertation was devoted to the origins of tastes and odors in municipal drinking waters. Dr. Lee has a bachelors degree in Environmental Science from San Jose State College, San Jose, California; a masters degree in Public Health from the University of North Carolina School of Public Health; and a Ph.D. degree from Harvard University, Cambridge, Massachusetts in environmental engineering and environmental sciences. Prior to coming to UTD, he taught and conducted research in the water supply-water pollution control field at the School of Public Health at the University of Pittsburgh, University of Wisconsin-Madison, and Texas A&M University. He has had approximately 20 years of teaching and research experience in the municipal and industrial water supply and water treatment field.

He currently serves as chairman of the American Water Works Association Committee on Quality Control in Water Supply Reservoirs. Professor Lee has published over 70 professional papers devoted to the area of eutrophication causes and control. G. Fred Lee has been involved for many years in the eutrophication control efforts in the Great Lakes states. He served as a member of the Eutrophication Committee of the Research Advisory Board for the International Joint Commission for Water Quality in the Great Lakes. He is currently a member of the IJC's Research Advisory Board Expert Committee on Engineering and Technology for Great Lakes Water Quality. Many of the opinions expressed in this report are the direct result of his experience in working on eutrophication control problems associated with the Great Lakes and in other parts of the US and world.

Elaine Meckel

Elaine Meckel received her bachelors degree in Marine Biology from the University of California at Berkeley in March 1976. She obtained a masters degree from the University of Texas at Dallas in August 1977. She is currently working toward the Ph.D. degree under the supervision of Dr. G. Fred Lee. Her Ph.D. dissertation is devoted to assessment of algal response relationships in Lake Ray Hubbard. In addition to authoring many of the initial drafts of many sections of this report, E. Meckel also served as project coordinator during the final phases of the project and final report.

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Walter Rast received his bachelors degree in Zoology from the University of Texas at Austin. He received a masters degree in Molecular Biology from the University of Texas at Dallas and a masters degree in Environmental Sciences from UTD. He is currently working for a Ph.D. degree under the supervision of G. Fred Lee on OECD eutrophication modeling. He currently holds a position as a member of the staff of the US-Canada International Joint Commission for Water Quality in the Great Lakes in Windsor, Canada.